

Understanding the effects of local and landscape variables on vegetable farm arthropod
communities

by

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B.S., South Dakota State University, 2023

A THESIS

submitted in partial fulfillment of the requirements for the degree

MASTER OF SCIENCE

Department of Entomology
College of Agriculture

KANSAS STATE UNIVERSITY
Manhattan, Kansas

2026

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Abstract

Urbanization is one of the main contributing factors to insect declines globally. To mitigate the losses of arthropods and other organisms, urban green spaces have been incorporated into city landscapes. Urban agriculture (UA) is a form of green space that has gained significant attention in the last decade. UA includes community gardens, vegetable farms, green roofs, home gardens, and many other similar land uses. This type of green space has been promoted as a beneficial habitat for arthropods. However, it is unclear how local and landscape variation across farms and gardens affects these green spaces' capacity to harbor diverse arthropods. Comparisons of habitat efficacy are also lacking between farms and other forms of green space, especially natural areas. Few studies examine the arthropod community broadly and encompass a range of feeding guilds and foraging strata. We addressed these knowledge gaps by examining the effects of landscape (developed cover) and local variables (plant richness, farm size, and habitat type) on arthropod groups (ground herbivores, foliar herbivores, ground predators, foliar predators, and pollinators) at vegetable farms practicing organic growing techniques and restored prairies in two field studies. The objectives of this project were to evaluate how local and landscape variables affect (A) the abundance and community composition of various arthropod guilds and (B) the community stability and foraging behavior of pollinators.

Our first study found that local variables were more influential than landscape variables in shaping the abundance and composition of ground herbivores, foliar herbivores, and foliar predators. However, we did find that in some cases abundance and composition were affected by interactions between local and landscape variables. Specifically, developed cover mediated the effects of crop richness on the abundance and composition of pollinators and the composition of

foliar herbivores. Additionally, farms and prairies supported similar types and quantities of pollinators and similar quantities of ground-dwelling predators. We also found that urban and peri-urban farms supported significantly more foliar predators than prairies. These findings support the claim that vegetable farms are favorable habitats for beneficial arthropods. Our second study examined the pollinator community more closely and used a plant-pollinator network approach to investigate community stability and foraging behavior. Pollinator species composition differed across urban and rural farms as well as between rural farms and prairies. However, no significant differences in community stability (network modularity) or foraging behavior (network specialization) were observed. Depending on conservation goals, prairies and some vegetable farms may be interchangeable habitats for pollinators. For example, if the purpose is to restore ecosystem function (i.e., stability, plant visitation patterns) to a landscape, these habitats may be equivalent. However, if the goal is to provide habitat for certain rare pollinators, these habitats may not have the same value.

These two studies showed that farms, and specifically UA, can be a beneficial habitat for arthropods of various feeding guilds. Farms may also support similar or greater quantities and types of certain arthropods compared to prairies. Incorporating more UA as green spaces in cities may help reduce local arthropod declines, as these spaces do harbor beneficial insects. As urbanization continues to change the landscape globally, retaining green spaces like UA will be important for promoting and maintaining ecological diversity.

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Acknowledgements

This project quite literally would not have been possible without the support and contributions of a whole village of people. I first want to thank Dr. Tania Kim for her unwavering support, encouragement, and guidance throughout this project. I also want to thank Dr. Brian Spiesman and Dr. Cary Rivard for their valuable insight and feedback. Thank you to the members of the Kim and Spiesman labs for help in the field and lab, feedback in the conference room, and moral support. This project wasn't easy, but they made the load a lot lighter. I want to recognize all the lovely people who make up the K-State Department of Entomology. I have learned so much from them and have deeply valued being a part of the tight-knit community they have formed. I would be remiss not to thank all the farmers and gardeners who took a chance on me, and welcomed our team into their places of work, and in some cases their backyards. Thank you to all my friends on and off campus who made Manhattan feel like home, even if it was just for a short while. Finally, I want to say thank you to my family and all of my friends who kept me sane even when my world seemed to be spinning out of control, and who believed in me even when I didn't believe in myself. Your texts, letters, phone calls, and visits meant more than you'll ever know.

Dedication

Above all, to God, my Hope and Redeemer, the one true source of peace I've found, and the only reason I do not worry about what tomorrow holds.

Introduction

Over the last decade, the warning bells have been rung, spreading word of a contemporary “insect apocalypse”: global arthropod declines reported by scientists and journalists alike (Goulson, 2019; Hallmann et al., 2017; Jarvis, 2018; Wagner et al., 2021). This is especially concerning as insect populations are often used as indicators of ecosystem health and resilience (Chowdhury et al., 2023; Yadav & Mishra, 2013). Many variables have been identified as contributing to arthropod declines (Wagner et al., 2021), including land use change, habitat connectivity, pollution, climate change, and urbanization (Eggleton, 2020; Sánchez-Bayo & Wyckhuys, 2019). These variables contribute to arthropod declines by various modes of action; they may cause direct mortality, disrupt life cycle activities (including reproduction), or increase stress (which may have repercussions on both reproduction and lifespan). For example, light pollution disrupts activity cycles, which can hinder mate-finding, or mating in general, of nocturnal arthropods, which may lead to lower reproductive rates (Bird & Parker, 2014; Owens & Lewis, 2022). Climate change is predicted to exacerbate phenological mismatches, where arthropods may respond to environmental cues and emerge earlier or later than when their food sources are available (e.g., plants or other arthropod prey), causing greater competition for resources and greater mortality (Renner & Zohner, 2018). For example, one study identified two species of lepidopterans in California that already struggle to match their lifecycles to the lifecycles of their host plants (Singer & Parmesan, 2010). Some larvae of the winter moth emerge before the host plant sets buds, and some larvae of Edith's checkerspot butterfly emerge after their host plant has already senesced for the season. Both of these situations can result in larval death via starvation (Singer & Parmesan, 2010). Singer and Parmesan also noted that climate change will likely alter the phenologies of these arthropods and their host plants, which

could result in better phenological alignment, but also has the potential to worsen these mismatches (2010).

Effects of urbanization on insects

Urbanization is a growing ecological concern for arthropods, as urban intensification and city expansion are both expected in the coming decades (Vaz et al., 2023). In 2025, the United Nations reported that 45% of the global population lived in cities; by 2050, they project this number will rise to 66% (United Nations, 2025). Urbanization is also a unique threat to arthropods because it is a combination of several factors. Researchers often partition these factors into two scales: landscape and local. Landscape factors influence the characteristics of the environment surrounding a site, shaping its broader habitat context. For example, two major variables of concern are city expansion and land use conversion. These processes result in the loss and fragmentation of habitat, which ultimately reduce/restrict resource availability (Gibb & Hochuli, 2002; Piano et al., 2020; Rebele, 1994; Ren et al., 2023). For example, habitat fragmentation has been shown to reduce pollinator visitation to plants (meaning a decline in foraging), and reduce pollination rates and seed set (Rathcke & Jules, 1993). Greater levels of fragmentation can also hinder gene flow and increase inbreeding (Kwak et al., 1998; Storfer et al., 2010). One study found that contemporary land use is one of the primary dispersal barriers hindering gene flow in *Bombus vosnesenskii*, an important pollinator in the southwestern United States (Jha, 2015). Similarly, this pattern was observed in a species of flightless ground beetle, where beetles in smaller habitats isolated by highways had lower genetic diversity (Keller & Largiadèr, 2003).

Local factors describe the immediate conditions, or quality, of a site. At this scale, factors of concern include pollutants (such as air, water, soil, sound, light, and thermal pollutants), habitat patch size, resource availability and quality (e.g., floral diversity and abundance), land management practices (e.g., herbicide and pesticide application, frequent mowing), and exotic plants (Collins et al., 2024; Jones & Leather, 2013; McIntyre, 2000). For example, a large quantity of exotic plants in an area influences the herbivore guild, and thus also strongly shapes the composition of predator and parasitoid species (Helden et al., 2012; Litt et al., 2014; Tallamy et al., 2021). One study found that non-native trees tended to support fewer arboreal hemipterans and insectivorous birds, and had lower hemipteran species richness than native trees (Helden et al., 2012). Patch size also influences community composition and species richness. Larger patches tend to have greater habitat heterogeneity, can support larger populations, and are more likely to be colonized by migrating arthropods (Faeth & Kane, 1978; Fattorini et al., 2018).

Urban green spaces and urban agriculture

Urban green spaces such as parks, forests, and gardens often mitigate both local and landscape-level issues found in urban areas. They do so by reducing urban heat island effects (C. Wang et al., 2022), remediating urban air pollution (Abhijith et al., 2017), reducing noise pollution (Ow & Ghosh, 2017), and providing habitat for wildlife and arthropods (Gallo et al., 2017; McIntyre, 2000; Prihandi & Nurvianto, 2022). These patches have also been found to benefit the health of city residents (Lee & Maheswaran, 2011; Tzoulas et al., 2007). For example, greater quantities of and closer proximity to green spaces were associated with better mental health and lower levels of anxiety, depression, and stress (Beyer et al., 2014; Sturm & Cohen, 2014; Sugiyama et al., 2008). Additionally, interacting with green spaces in cities has

been linked to increased lifespans in senior citizens (Takano et al., 2002), and a reduction in mortality (Villeneuve et al., 2012).

Urban Agriculture (UA) is one form of urban green space that has received a lot of attention in the last decade or so and became especially prominent in response to the COVID-19 pandemic (Pinheiro & Govind, 2020; Schoen et al., 2021). UA is defined as the practice of growing/raising, processing, and distributing food or non-food products within towns or cities (FAO, 2022). This practice is far from new, however, as remains of agricultural sites have been found within ancient cities like Pompeii and Babylon (Jashemski, 1992; Orsini et al., 2020). Food was historically grown in cities because it was difficult to transport produce, grains, and meat over long distances from rural to urban areas and maintain their quality (Barthel & Isendahl, 2013; Doolittle, 2004). With advances in technology and agriculture, the long-distance transportation of food became easier, and large-scale agriculture became more efficient (Barthel & Isendahl, 2013; Burgin, 2018). This ultimately encouraged the rural production of food and made UA less popular. Resurgences of the popularity of UA in the United States have been seen many times throughout history in times of war and as parts of civic and environmental movements (Hynes & Howe, 2004; Lucas, 2025; Mok et al., 2014). In the United States, UA is less strongly tied to household nutrition and income than it is in developing countries (Senthamizh & P, 2025). Rather, it is often associated with goals of environmental sustainability, food system resilience, social equity, and education (Mok et al., 2014; Senthamizh & P, 2025). Common forms of contemporary UA include rooftop gardens, community gardens, commercial farms, backyard gardens, and institutional gardens. These types of green spaces are often acknowledged for their multiple-use capacity, as these sites produce food sustainably, provide

health benefits, engage and educate the public, and function as habitat for arthropods and other wildlife (Lovell, 2010; Orsini et al., 2020).

Importance of arthropods in urban agriculture

Several studies examine the effects of urbanization on arthropods; however, much is unknown about the ability of farms and gardens specifically to support arthropod communities along an urbanization gradient (Bell et al., 2023; Collins et al., 2024). This knowledge gap is important to fill as urban farms and gardens are a unique study system, and arthropods provide vital ecosystem services which many agricultural sites heavily rely on. These ecosystem services include things like pollination, pest control, soil fertility, and carbon sequestration, which are all vital to growing healthy plants and producing fruits and vegetables (Haan et al., 2021; Isaacs et al., 2009). Thus, it is imperative that we understand how arthropod communities respond to challenges like urbanization on farms and gardens, specifically to determine how best to support them for the sake of maintaining these ecosystem services, but also to mitigate global biodiversity losses.

Pollinators

Pollinators are one group of insects that are vital to the stability of our global food system. Seventy-five percent of global food crops rely on arthropod or animal pollination (Potts et al., 2016). UA has been identified as a potential hotspot for pollinators, and this form of green space may help moderate local pollinator declines by providing suitable habitat and enhancing biodiversity (Baldock et al., 2019; Theodorou et al., 2020). UA has specifically been found to better support pollinators than other land uses, including city parks, road verges, and even

traditional rural agriculture (Baldock et al., 2019; Kaluza et al., 2016; Olsson et al., 2021).

However, it is unclear if farms and gardens can support similar quantities and types of pollinators as prairies, which are known to be high-quality habitats. Studies suggest that natural areas like bushland and forest support different pollinator species than gardens (Marín et al., 2020; Prendergast & Ollerton, 2021), but comparisons between farms/gardens and prairies have not been reported. Additionally, most studies focus on pollinator abundance and richness, but do not consider how environmental variables may influence foraging behavior or pollinator community structure. Alterations to foraging behavior may pose a threat to the pollination success of crop plants (Andrieu et al., 2009; Revilla & Křivan, 2016).

Bipartite networks can be used to assess patterns in plant-pollinator interactions like these. For example, modularity, a common network metric, is used to describe the structure of networks and can give insight into their resistance to disturbance (Newman, 2006).

Specialization is another common network metric and is used to study the foraging behavior of pollinators by examining which plant species, and the number of different plant species that pollinators visit (Blüthgen et al., 2006). From a conservation standpoint, these metrics can also be used to inform seed mixes for habitat plantings (Elle et al., 2012). Understanding the effects of urbanization on network structure in urban farms will help clarify the quality of these places as habitat for pollinators. It will also provide insight into whether these places only support common generalist pollinator species or if rarer species are also able to benefit from these green spaces.

Thesis objectives

The two primary goals of this thesis were to:

- 1) Understand how developed cover, farm size, crop richness, and habitat type (farm vs prairie) drive arthropod abundance and community composition.
- 2) Understand how farm size, crop richness, and habitat type (farm vs prairie) influence pollinator foraging behavior, community stability, and species assemblages.

Both objectives were addressed through observational field studies. These studies also compared arthropod communities between farms and prairies to determine habitat suitability. For my first objective, I conducted a two-year field study that investigated the effects of developed cover, farm size, crop richness, and habitat type (farm vs prairie) on the abundances and community compositions of arthropods from different guilds and foraging strata. For my second objective, I performed a one-year observational field study of plant-pollinator interactions to specifically study the effects of habitat type (urban farm, peri-urban farm, rural farm, or restored prairie), site size, and floral richness on pollinator behavior and community structure. Together, these results will give insight into the effects of urbanization on arthropods, as well as the value of vegetable farms as arthropod habitat.

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Chapter 1 - Effects of local and landscape variables on the abundance and community compositions of pests, predators, and pollinators

Abstract

Globally, insect populations are in decline. Several variables have been identified as causes for these declines, one of the predominant factors being urbanization. To combat the effects of urbanization on arthropods and other wildlife, green spaces have been introduced to cities. One specific form of green space, urban agriculture (UA), has been steadily increasing in popularity over the last decade. UA includes spaces like community, backyard, and educational gardens, and market farms. This form of green space has been suggested to be beneficial habitat for arthropods; however, its quality compared to other types of habitat has not been well studied. It is also not well understood how arthropods from different feeding guilds and foraging strata at gardens and farms respond to various local and landscape factors associated with urbanization. To address these knowledge gaps, we examined the effects of developed cover, farm size, and crop richness on arthropod abundance and community composition at a series of vegetable farms. We also compared arthropod abundances and compositions between farms and prairies. Our study found that arthropods from different foraging strata were predominantly influenced by different variables. Crop richness was an important local variable that affected the abundance and composition of several arthropod feeding guilds. We also found that farms supported similar quantities and types of ground-dwelling predators and pollinators compared to prairies. Urban and peri-urban farms also supported similar significantly more foliar predators than prairies. Our results affirm the claim that vegetable farms are valuable habitats for beneficial arthropods.

Preserving UA and introducing more similar types of green spaces in cities may help reduce local arthropod declines and protect ecological diversity.

Introduction

Urbanization is one of the driving factors reducing arthropod diversity and abundance globally (Sánchez-Bayo & Wyckhuys, 2019; Fenoglio et al., 2020; Raupp et al., 2010). Habitat fragmentation and direct loss of habitat are two ways that urbanization affects arthropods (Vaz et al., 2023). Habitat fragmentation refers to a habitat being divided into several smaller pieces, while habitat loss refers to a reduction in total area in the landscape (Faaborg et al., 1995; Fahrig, 2003). Habitat fragmentation hinders arthropod dispersal, creating more isolated populations and an increased risk of inbreeding (Ho et al., 2025; Jha & Kremen, 2013). As a result, fragmentation typically leads to smaller arthropod populations and less diverse communities (Hunter, 2002). Urbanization also contributes to land use change, which causes direct habitat loss. This shrinks the size of habitats, which reduces resource abundance and generally causes losses in arthropod abundance and richness (MacArthur & Wilson, 1967; With & Pavuk, 2011).

Not all arthropod taxa respond to urbanization in the same way; some are more sensitive to fragmentation and habitat loss than others (Jones & Leather, 2012). Certain arthropods may also be able to exploit resources that are more common in urban areas. For instance, sap-feeding insects seem to benefit from urbanization, as they are able to exploit ornamental plant resources in cities, while their natural enemies are more sensitive to habitat fragmentation and show declines (Fenoglio et al., 2020; Korányi et al., 2022; Kruess & Tschamntke, 1994). Researchers disagree on the habitat quality of urban areas for certain beneficial arthropods, including pollinators. Some reviews find that urban areas may be a favorable habitat for bees (Hall et al.,

2017; Prendergast et al., 2022), while others label urbanization a threat to this group (Liang et al., 2023; Millard et al., 2021). Clearly, more research is needed to parse out the driving factors behind these discrepancies and understand arthropod trends across urbanization gradients as we try to slow global declines (Sánchez-Bayo & Wyckhuys, 2019).

Green spaces have been introduced in many cities to mitigate urbanization's effects on biodiversity (Lepczyk et al., 2017; Sadler et al., 2010). Urban agriculture (UA) has become an increasingly popular form of green space, and includes diversified market farms, community and/or educational gardens, and home gardens. UA is valued for being multiple-use as it can produce food sustainably, stimulate community engagement, and provide habitat for organisms in the midst of an inhospitable landscape (Di Pietro et al., 2018; Gaston & Gaston, 2010).

However, not all UA sites are created equal; they exist in a wide range of sizes, from small corners of properties to multiple city blocks. Depending on their size and goals of production, gardens may grow a large variety of vegetables and herbs or specialize in a few different types of plants. UA also exists across a gradient of urbanization; gardens and market farms may be found in the densely populated urban core, or near less dense areas.

Each of these variables (size, planting diversity, and degree of urbanization) has the capacity to influence the biota UA can support. Larger sites/habitat patches will generally have more diversity and support greater quantities of arthropods than smaller ones (MacArthur & Wilson, 1967). Greater plant richness also promotes greater arthropod richness (Siemann et al., 1998). Finally, arthropod abundance and richness decline with greater urbanization (Vaz et al., 2023).

If UA is going to be promoted as a harbor for biodiversity, more research is needed to quantify its capacity to support diverse arthropod assemblages compared to other forms of green

space. Studies show that urban green spaces, including UA, may better support certain arthropod groups like pollinators than rural agriculture, but tend to be inferior to natural areas (Wenzel et al., 2020). Urban areas also tend to better suit generalist and opportunist species, though these trends are not consistently observed (Liang et al., 2023; Wenzel et al., 2020). Additionally, research shows that gardens and urban farms can act as stepping stone habitats to facilitate the movement of arthropods and other organisms through the landscape (Egerer et al., 2020; Goddard et al., 2010; Langellotto et al., 2018). However, it is unclear how arthropod communities in UA compare to those in restored prairies in the midwestern United States. UA may not have the same conservation value if it cannot support rare species that natural areas can.

In this study, we explore how landscape and local factors influence arthropod communities on market farms and gardens practicing organic growing techniques along an urbanization gradient in the Central United States. We also compared these agricultural sites to restored prairies as a baseline for arthropod abundance and to understand the relative quality of UA as a form of arthropod habitat. The driving questions for this study are as follows:

- (1) How do landscape (surrounding developed cover) and local variables (farm size and crop richness) influence arthropod abundance and composition?
- (2) Will arthropods of similar diet types (predator vs herbivore) or foraging location (ground vs foliar) respond to landscape and local variables in similar ways?
- (3) Do farms support similar quantities and types of beneficial arthropods compared to restored prairies?

We hypothesize that groups of the same foraging location (ground vs foliar) will be more similar in the factors they respond to than guilds of the same diet type (pest vs natural enemy). We hypothesize that the amount of surrounding developed cover in the surrounding landscape

will be more influential in shaping the abundance and community composition of foliar and pollinating arthropods, and that crop richness and farm size (both local factors) will be most influential in shaping the abundance and composition of ground-dwelling arthropods. We build these predictions upon the idea that foliar arthropods tend to have greater foraging distances and dispersal ability than ground-dwelling arthropods and thus interact more frequently with the greater landscape (Tscharrntke et al., 2005). As a result, we expect ground-dwelling arthropods will primarily be affected by the local resources that they have at their disposal. These organisms will be subject to the same microenvironments and have more similar dispersal capabilities to each other (Roper-Edwards & Hurlbert, 2024). Finally, we predict that restored prairies will have more natural enemies and pollinators and fewer pests than farms. We also expect arthropod community compositions to be different between restored prairies and farms.

Methods

A combination of 19 market farms, school farms, community gardens, and church gardens were selected as study sites (Fig. 1.1). For the purpose of this report, all agricultural locations will be referred to as “farms”. These farms all reported using organic growing practices, though few were organic certified. Fourteen farms were selected for the first field season (June to August 2024), and twelve were selected for the second field season (June to August 2025). Of the farms visited in 2025, seven were repeat sites from 2024, and five were new sites. Additionally, three restored prairies were selected for the 2025 field season (Fig. 1.1). These sites provided a baseline comparison for the ability of farms to support arthropods. Not all farms grew the same fruits, vegetables, herbs, or ornamental flowers, so common crops were selected for transects when possible. Each site was visited twice per month for the three months

of the study (June, July, and August), resulting in six sampling dates per site. A variety of techniques were used to characterize the arthropod community within each transect, including pitfall traps, yellow sticky traps, 10-meter sweep net samples, and 10-minute timed floral observations (see full descriptions below).

Site selection

Study sites were located in Northeastern Kansas and Northwestern Missouri and generally fell within the Kansas City metropolitan region. All farms grew a variety of fruits, vegetables, herbs, and ornamental flowers. Management strategies and production intensity differed based on the size of the property and whether their produce was sold to the public. Most farms lacked formal organic certification, but all sites reported using organic growing practices. Sites were selected along a gradient of urbanization, ranging from urban to rural. This gradient was determined using ArcGIS Pro and the USDA NASS Cropland Data Layer (USDA NASS, 2024). A 1 km buffer around the center point of each farm was used to determine the degree of urbanization by calculating the percentage of developed land class within the buffer. We used this buffer size because it is an average distance of movement for the arthropods examined (Egerer et al., 2017). Developed land was considered the sum of “high”, “medium”, “low”, and “open space” developed classes from established Cropland Data Layer categories.

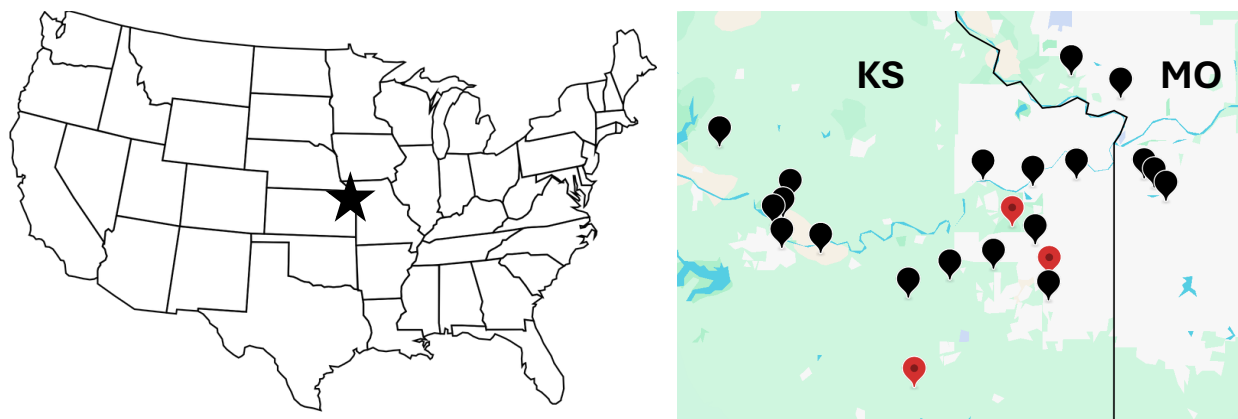


Figure 1.1. Map of study sites within the Kansas City metropolitan area. Black pins represent farm sites and red pins represent prairie sites.

Several metrics have been used as urbanization proxies across the literature (Sanllorente et al., 2025). From each site's center, we measured road length within a 1 km buffer, distance of site to city center, proportion of natural area within a 1 km buffer, and various combinations of developed cover (high, medium, and low intensity, and open developed area) within a 1 km buffer. Upon completion of a correlation test, all of these metrics were highly correlated. Therefore, we opted to use the total proportion of developed cover (high, medium, low, and open intensities all added together) within a buffer with a radius of 1 km as our urbanization proxy because this was the metric we initially used for site selection, and it resulted in the most evenly distributed range of values. Our sites ranged from 4% to 100% total developed cover within this buffer size.

Farms were also chosen to encompass a range of sizes. Farm size was measured using Google Earth by drawing polygons around the cultivated portion of the site and calculating the area. Crop richness was measured for each farm by recording the different types of vegetables and herbs observed being grown each month in June, July, and August. These lists were compiled for each farm individually to produce one number representing the total count of different vegetable and herb crops grown across the summer.

Prairie sites were selected in the same way as the farms, across a gradient of size and urbanization within the KC metropolitan area. All prairies were managed by the Johnson County Park and Recreation District. They were restored at least five years prior to this study and were seeded with mixes of forbs and grasses. The prairies were not mowed, burned, or sprayed throughout the duration of this study.

Arthropod collection

At each site, 10 m transects were established (N=4 in 2024, N=3 in 2025). These transects were positioned within rows of fruit and vegetable crop plants, herbs, or ornamental flowers (or among wildflowers at the prairie sites). Transects were as evenly spaced across the farm as possible, while also encapsulating crop richness for the farm. At the first visit of each month, one pitfall and one sticky trap were placed in the center of each transect, and timed observations were conducted if the transect was flowering. One week later, the second visit of the month took place. Pitfall and sticky traps were collected, 10 m sweeps were performed, and timed observations were conducted again if the transect was flowering.

Pitfall traps

Pitfall traps were used to survey the ground-dwelling arthropod community and consisted of one 0.5 L Solo Cup submerged belowground so that the rim of the cup was level with the soil surface. Each cup was filled half-full of a mixture of soapy water and covered with a 20 x 20 cm square of wire mesh with 3 cm diameter openings to prevent vertebrates from entering the trap, which was fixed with metal garden staples. Each trap also had a 20 x 20 cm square of white corrugated plastic held approximately 5 cm above the soil surface to prevent rainwater from

flooding it. Traps were left in the field for approximately seven days before retrieval. Arthropods in the cups were strained from the soapy water mixture in the field and stored in sample containers with 90% ethanol at room temperature for preservation. Arthropods were identified to taxonomic order or family groups of interest (see categories in Table 1.1) using a microscope.

Sticky traps

The flying insect community was surveyed using 23 x 28 cm yellow Pherocon AM No-Bait Sticky Traps. At the center of each transect, one trap was tied to a metal pole at approximately the same height as or slightly above the surrounding plants. Traps were deployed for approximately seven days before being collected. Upon retrieval, clear cellophane sheets were used to cover the sticky portion of the trap to allow for easier processing. Traps were transported from the field to the lab inside a cooler and stored in a -20° C freezer until the time of processing. Insects were primarily identified to broad taxonomic groups such as dipterans, hemipterans, and lepidopterans, except for groups of interest (e.g., aphids, lady beetles, long-legged flies, etc.; see Table 1.1 for categories). Thrips and whiteflies were grouped together when classified. Most traps contained large numbers of winged aphids, thrips, and whiteflies. To reduce processing time for these insects, we subsampled individuals within 1/3 of the grid squares printed on each trap in a consistent pattern, counting four squares from each corner and four from the center (total 20 squares), and extrapolated counts by multiplying this sub-count by three. A correlation test revealed that the extrapolated count was significantly highly correlated with the true total count ($r=0.996$, $p<0.001$; Fig. 1.2), and thus made a reasonable substitute.

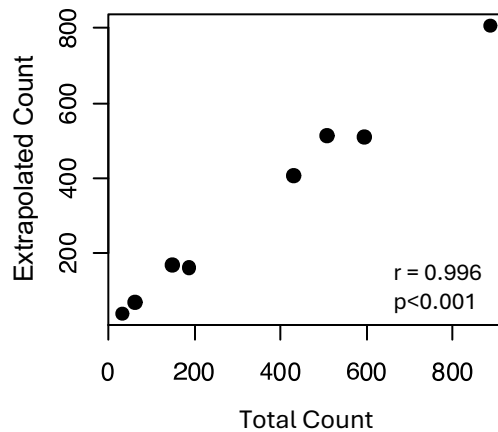


Figure 1.2. Correlation between true and extrapolated aphid counts ($r=0.996$, $p<0.001$).

Sweep net samples

To sample insects that live and forage on vegetation, sweep sampling with 36-cm-diameter aerial nets was performed. Gentle sweeps through vegetation were used to not damage high-value crops while still being vigorous enough to prevent insects from escaping the net. Each sample was standardized by length (approximately 10 meters). Samples were stored in Ziplock bags in a -20°C freezer until the time of processing. Arthropods were identified primarily to the family level and then stored in 90% ethanol at room temperature.

Timed observations

Pollinators were sampled via 10-minute timed observations where arthropods were collected off flowers using hand-held insect vacuums (modified cordless Skil hand vacuums). Transects could be sampled up to twice per month if they were flowering. Some transects never flowered and thus were never sampled for pollinators. After capture, pollinators were transferred from the vacuums to 120 ml plastic sample cups and then stored in a -20°C freezer until the time of processing. Bees and syrphid flies were pinned and identified to family level or lower (see Table 1.1).

Data organization

After arthropods were identified to the corresponding family or order level, all samples were pooled for a single site per year. The data were then standardized by dividing each family/order abundance value by the number of transects/samples that had been pooled. For example, in 2024, there were four transects, each with a sticky trap; traps were set out three times over the season, for a total of 12 traps. Thus, the abundance values of each taxonomic group were divided by 12. In the instance that a sample from a site was missing, the divisor reflected the true number of samples that were processed and pooled together. Pollinator abundance was standardized by the number of potential samples that could have been taken at that site (i.e., the number of transects multiplied by six: two visits each month for three months). Even though samples may not have been collected at a transect, we used this method to more accurately capture the pollinator abundance at that farm. If few transects were flowering at a given site, then it was likely that the farm provided fewer total floral resources and thus supported fewer pollinators overall.

Once the data were standardized, each family/order was assigned to a guild (Triplehorn, 2005). These were ground-dwelling pests (ground pests), ground-dwelling natural enemies (ground predators), foliar pests, foliar natural enemies (foliar predators), and pollinators. Not all arthropods fit absolutely into these categories, so some were excluded from analyses (i.e., most dipterans). Though isopods are traditionally considered detritivores, they are also known to damage seedlings of various vegetable species and feed on garden fruits (Messelink & Bloemhard, 2007). Therefore, we categorized them as ground pests for our study. After this process had been completed for each collection method individually, all collection methods for a

single site were pooled.

Statistical analysis

To analyze abundance, the adjusted abundance totals by site for each of the five groups were calculated and log-transformed to meet normality. Thus, five response variables were tested: log(ground pest abundance), log(ground predator abundance), log(foliar pest abundance), log(foliar predator abundance), and log(pollinator abundance). We used linear models and backwards stepwise model regression. The step function with the specification “backwards = true” from the *stats* package in R was used for this (R Core Team, 2024). Models were fit individually for all five of the previously listed response variables. Our predictor variables included collection year, the proportion of developed cover within a 1 km buffer (our proxy for urbanization), the number of different crops present at the site across the season (standardized as z-scores) the size of the site (measured in hectares, and also standardized as z-scores), and three two-way interactions: developed cover x crop richness, developed cover x site size, and crop richness x site size. Linear models for each response variable were initially fitted with all predictor variables. Unnecessary variables were systematically removed based on minimizing model AIC values until a best-fit model with the lowest AIC value was achieved. Variables were not removed based on their p-values, only whether their removal would lower model AIC. Interaction terms were visualized using the *effects* package (Fox & Weisberg, 2019).

To assess differences in community composition within each of the five groups, we used permutational multivariate analyses of variance (PERMANOVA) (*adonis2* function from the package *vegan*; Oksanen et al., 2025) and principal coordinate analyses (PCoA). The final PERMANOVA models used for analyses were achieved using a reverse selection method. First,

a model was fit with all predictor variables, and one at a time, the variable with the lowest R^2 was manually removed until all predictor variables in the model were significant. PCoA plots and PERMANOVAs were based on a Bray-Curtis dissimilarity matrix and 999 permutations. They were visualized using the package *ggplot2* (Wickham, 2016). Pairwise PERMANOVA was conducted for interaction terms using the package *pairwiseAdonis* (Martinez Arbizu, 2017). Point dispersion was also calculated using the function ‘betadisper’ from the package *vegan* and compared using pairwise PERMANOVA. All statistical analyses were performed in R statistical software (version 4.4.2; R Core Team, 2024).

To compare farm and prairie arthropods, we divided farms into three urbanization categories: rural (0-33% developed cover), peri-urban (34-66% developed cover), and urban (67-100% developed cover). ANOVA and Tukey’s HSD were used to compare abundances, and PERMANOVA and PCoA were used to compare community compositions.

Results

Over two years, we collected approximately 307,470 arthropods from the 22 sites. After categorizing them into our five groups, the final analysis included 247,609 arthropods (Table 1.1). Ground pests consisted of 27,848 (11.25%) specimens representing five taxonomic groups, over 80% of which were isopods. Ground predators consisted of 34,436 (13.91%) specimens from five taxonomic groups, over 80% of which were ants. Foliar pests consisted of 160,365 (64.77%) specimens representing 11 taxonomic groups, over 50% of which were thrips and whiteflies. Foliar predators consisted of 23,582 (9.52%) specimens representing 10 taxonomic groups, over 50% of which were wasps. Pollinators consisted of 1,378 arthropods (0.55%), and represented 13 taxonomic groups, 27% of which were sweat bees, 25% bumble bees, and 24%

honey bees. We found that the sampling year had significant effects on foliar pest community compositions ($F_{(1,19)}=20.29$, $p<0.01$; Table 1.3) and abundances ($\beta = -0.31$, $p<0.01$; Table 1.2). Specifically, more foliar pests were captured in the first year of the study (2024; Fig. 1.3).

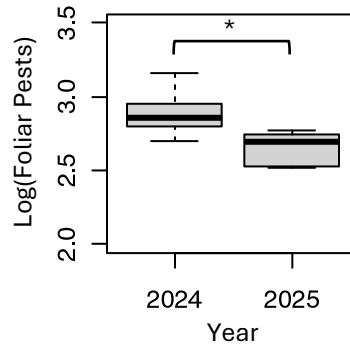


Figure 1.3. Boxplot showing differences in arthropod abundance between study years for foliar pests.

1 **Table 1.1. List of taxonomic classifications included in each of the following five guild groups: ground-dwelling pests, ground-**
2 **dwelling predators, foliar pests, foliar predators, and pollinators. Taxonomic groupings are listed in order of their respective**
3 **percent of group composition. Wasps were primarily parasitoids, with some predatory species.**

Ground Pests		Ground Predators		Foliar Pests		Foliar Predators		Pollinators	
Taxonomic Group	% Comp.	Taxonomic Group	% Comp.	Taxonomic Group	% Comp.	Taxonomic Group	% Comp.	Taxonomic Group	% Comp.
Isopoda	81.66	Formicidae	85.72	Thysanoptera &	56.20	Wasps (Parasitica & Aculeata)	57.54	Halictidae	27.33
Coleoptera	7.61	Staphylinidae	6.83	Aleyrodidae				<i>Bombus spp.</i>	24.60
Hemiptera	5.41	Araneae	4.55	Aphididae	25.87	Dolichopodidae	23.15	<i>Apis mellifera</i>	23.95
Orthoptera	4.84	Carabidae	2.70	Auchenorrhyncha	8.19	Formicidae	13.68	Syrphidae	7.40
Diplopoda	0.48	Chilopoda	0.20	Other Coleoptera	5.94	Coccinellidae	2.80	Other Apidae	5.88
				Other Hemiptera	1.46	Anthocoridae	1.66	<i>Ceratina spp.</i>	4.86
				Miridae	1.06	Neuroptera	0.44	Megachilidae	3.77
				Berytidae	0.46	Cantharidae	0.37	<i>Xylocopa virginica</i>	1.38
				Lepidoptera	0.38	Araneae	0.15	Colletidae	0.87
				Scarabaeidae	0.23	Lampyridae	0.14	Andrenidae	0.15
				Chrysomelidae	0.20	Predatory Hemiptera	0.07		
				Curculionidae	0.01				
Total	100		100		100		100		100

4

Arthropod abundances

Both local and landscape factors influenced arthropod abundances on urban farms. The amount of developed cover surrounding the farm, a landscape factor, had a positive association with the abundances of ground pests ($\beta=0.55$, $p=0.03$) and foliar predators ($\beta=0.13$, $p<0.01$; Table 1.2). Crop richness, a local factor, also increased ground pest abundances ($\beta=0.28$, $p=0.02$), but interacted with farm size to affect foliar pests ($\beta=0.19$, $p<0.01$) and predators ($\beta=0.08$, $p<0.01$). In particular, foliar pest abundance declined with crop richness but only on small farms (Fig. 1.4a). Conversely, foliar predator abundance increased with crop richness but only on large farms (Fig. 1.4b).

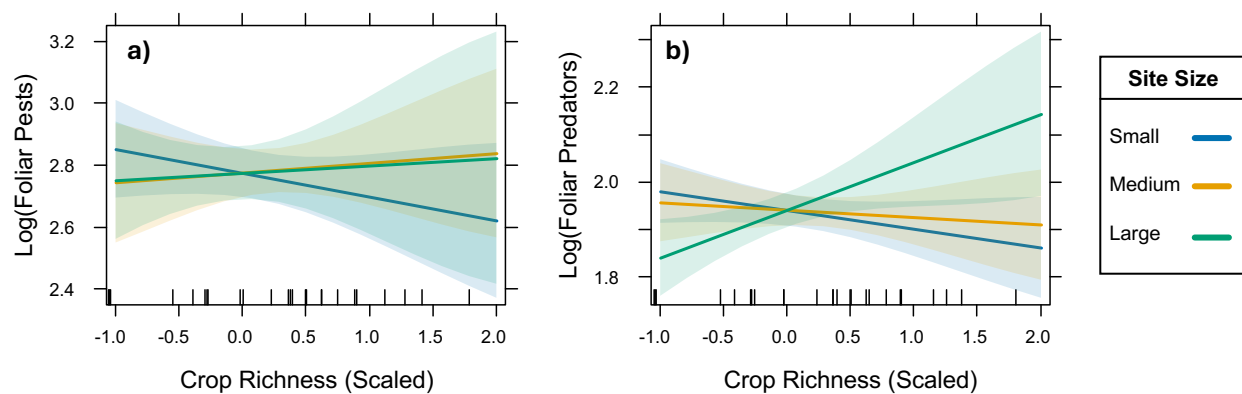


Figure 1.4. Interaction plots of crop richness and site size for the abundance of: (a) foliar pests, (b) foliar predators. Crop richness was scaled using a z-score transformation, and farm size was binned into three categories: small (0.1-0.8 ha), medium (1.3-1.8 ha), and large (2.3 - 9.5 ha).

Regarding pollinator abundance, we saw a significant interaction between a local and landscape variable (crop richness and developed cover; $p=0.02$). This interaction showed that an increase in crop richness reduced pollinator abundance, but only on urban farms (Fig. 1.5).

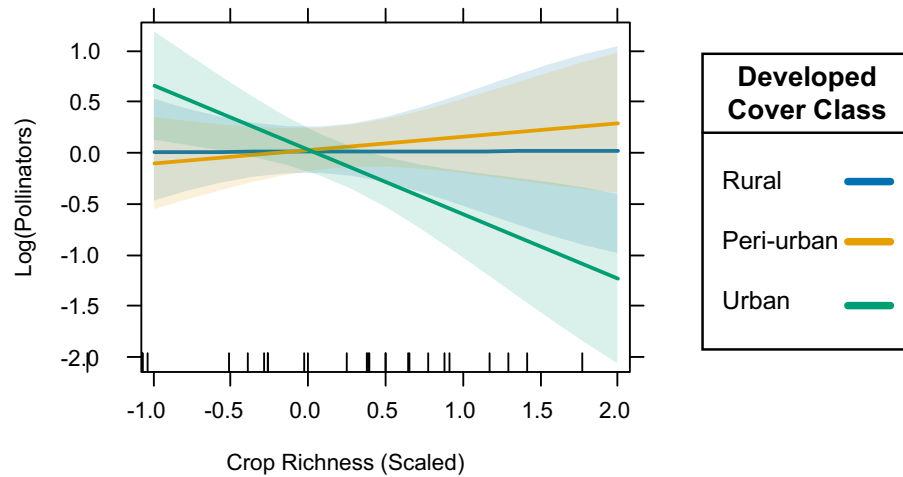


Figure 1.5. Interaction plot of crop richness and developed cover class for pollinator abundance. Crop richness was scaled using a z-score transformation, and developed cover was binned into three categories: rural (0-33% developed), peri-urban (34-66% developed), and urban (67-100% developed).

Table 1.2. Results of the AIC backwards stepwise model selection for the abundances of each guild group of interest. No β values are reported if the corresponding variables were not selected in the final model. Significant variables ($p \leq 0.05$) have been bolded and marked with an asterisk. Adjusted R^2 values and F-statistics are listed for each model.

Guild	Developed Cover	Dev. Cover x Crop Richness	Crop Richness	Farm Size	Crop Richness x Farm Size	Sample Year	Adjusted R^2	F
Ground Pests	$\beta=0.55$ $p=0.03^*$		$\beta=0.28$ $p=0.02^*$			$\beta=-0.28$ $p=0.13$	0.34	5.25 DF: 3, 22 $p<0.01^*$
Ground Predators	$\beta=0.43$ $p=0.13$						0.05	2.45 DF: 1, 24 $p=0.13$
Foliar Pests	$\beta<-0.01$ $p=0.96$	$\beta=0.23$ $p=0.09$	$\beta=0.02$ $p=0.66$	$\beta=0.13$ $p=0.01^*$	$\beta=0.19$ $p<0.01^*$	$\beta=-0.31$ $p<0.01^*$	0.64	8.31 DF: 6, 19 $p<0.01^*$
Foliar Predators	$\beta=0.13$ $p<0.01^*$		$\beta=0.04$ $p=0.08$	$\beta=0.06$ $p=0.04^*$	$\beta=0.08$ $p<0.01^*$	$\beta=-0.04$ $p=0.08$	0.48	5.60 DF: 5, 20 $p<0.01^*$
Pollinators	$\beta=0.26$ $p=0.38$	$\beta=-0.96$ $p=0.02^*$	$\beta=0.31$ $p=0.18$				0.14	2.34 DF: 3, 22 $p=0.10$

Insect community composition

Ground pest community compositions were affected by the amount of developed cover in the surrounding landscape ($F_{(1,24)}=4.45$, $p=0.01$; Table 1.3), where peri-urban farms supported fewer isopods and rural farms supported fewer orthopterans compared to the other farm types (Fig. 1.6a).

Table 1.3. Results of PERMANOVA tests on community composition. F-statistics and p-values of variables present in each best-fit model are reported. All values listed were statistically significant ($p \leq 0.05$) and denoted with an asterisk. No value is provided if the variable was not included in the final model. R^2 values are listed for each model. Ground predators and foliar predators had no significant variables remaining at the end of model selection.

Guild	Developed Cover	Developed Cover x Crop Richness	Crop Richness x Farm Size	Sample Year	R^2
Ground Pests	$F_{(1,24)}=4.45$ $p=0.01^*$				0.16
Ground Predators					-
Foliar Pests		$F_{(1,19)}=4.43$ $p=0.01^*$	$F_{(1,19)}=5.54$ $p<0.01^*$	$F_{(1,19)}=20.29$ $p<0.01^*$	0.64
Foliar Predators					-
Pollinators		$F_{(1,19)}=2.25$ $p=0.02^*$			0.09

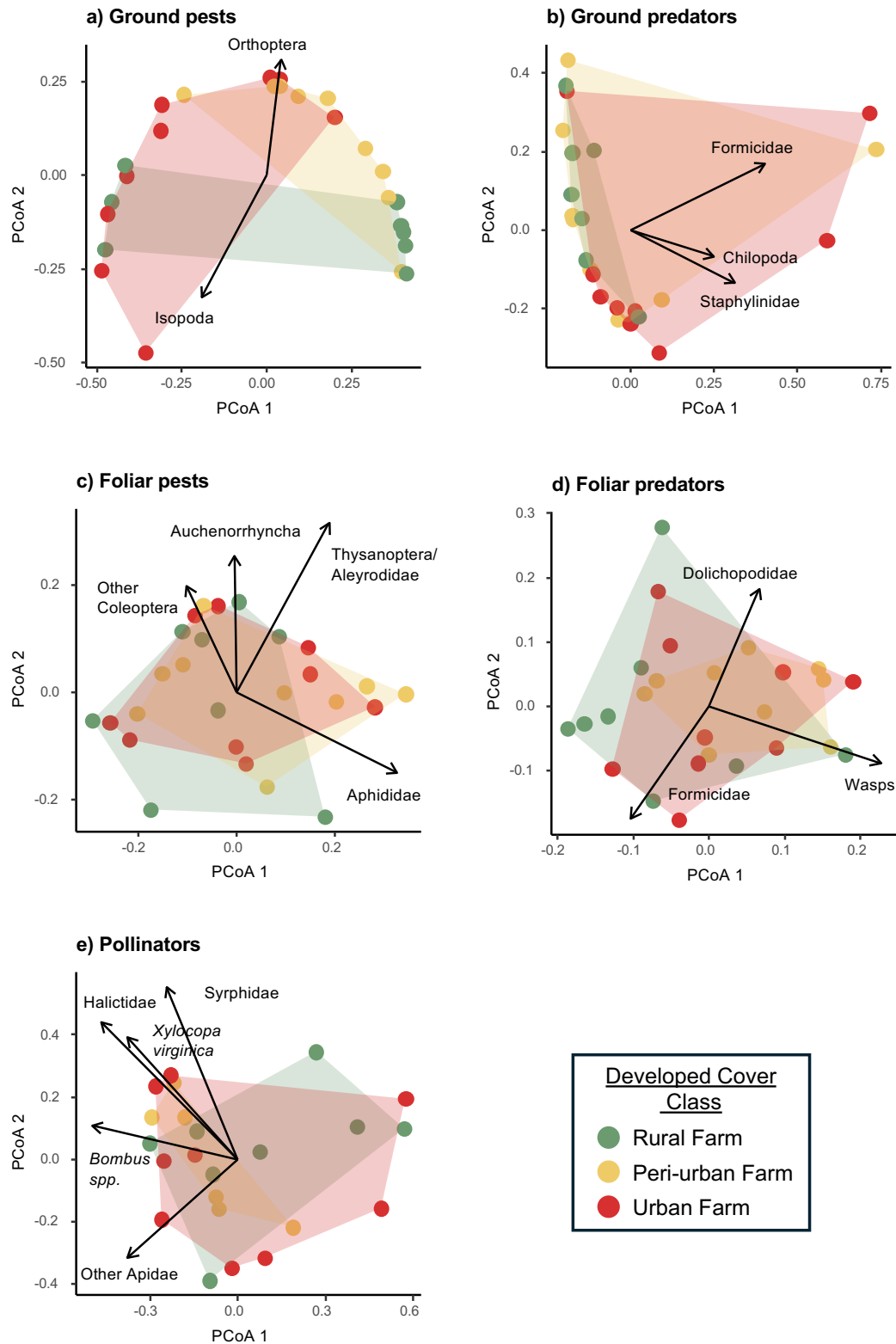


Figure 1.6. PCoAs for each arthropod guild group, including vectors of significant species: (a) ground pests, (b) ground predators, (c) foliar pests, (d) foliar predators, (e) pollinators. Sites are color-coded in ordination space by developed cover classes.

On the other hand, foliar pest compositions were affected by the interaction of developed cover with crop richness ($F_{(1,19)}=4.43$, $p=0.01$). Pairwise PERMANOVA showed a marginally significant difference in foliar pest community composition between rural farms at high versus low crop richness ($p=0.07$; Fig. 1.7a). Rural farms with low crop richness harbored more thrips, whiteflies, leafhoppers, and beetles compared to rural farms with high crop richness (Fig. 1.7).

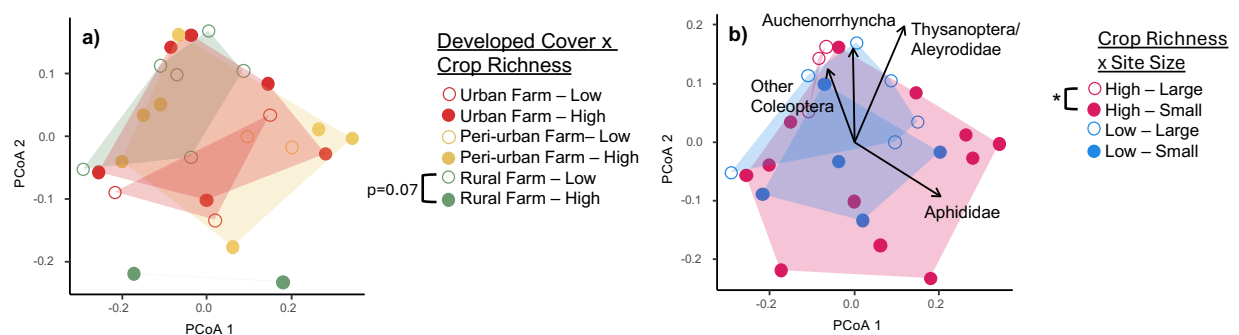


Figure 1.7. PCoAs of significant interaction terms for foliar pests: (a) developed cover and crop richness, (b) crop richness and site size. All variables were treated as continuous to plot points, but colored and assigned shapes using categorical versions. In plot a, open points represent low crop richness (3-14 unique crops) and closed points represent high richness (>16 unique crops). In plot b, blue points represent low crop richness and pink high richness; open points represent large sites (>2 ha) and closed represent small sites (<2 ha).

Foliar pest composition was also affected by the interaction of crop richness and farm size ($F_{(1,19)}=5.54$, $p<0.01$). Specifically, within farms with high crop richness, there was less variation in foliar pest community composition at large sites (i.e., these sites had similar pest communities) compared to small farms where pest community compositions were very different from farm to farm (point dispersion: $p=0.01$, Fig. 1.7b). Large farms with high crop richness tended to have more beetles and leafhoppers than small farms with high crop richness (Fig. 1.7b).

Pollinator community composition was also affected by an interaction between developed cover and crop richness ($F_{(1,19)}=2.25$, $p=0.02$; Table 1.3). However, only a marginally significant difference was found, which was between rural farms of high and low crop richness ($p=0.06$; Fig. 1.8). The compositions of predator communities (ground and foliar) were not significantly affected by any of the variables we examined.

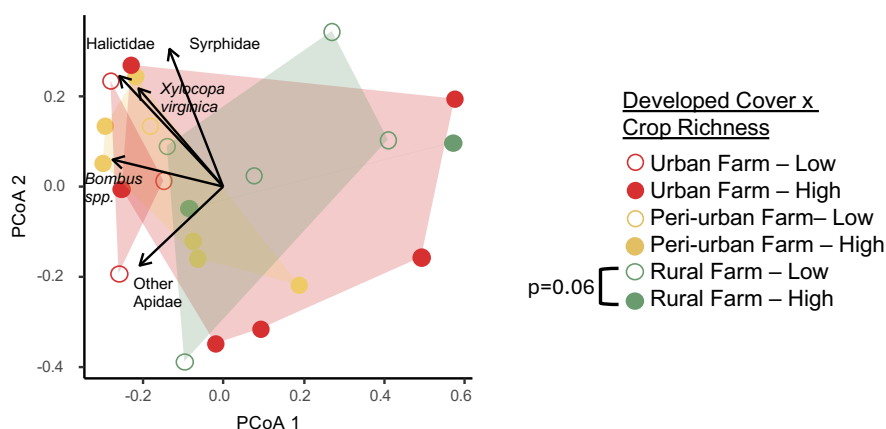


Figure 1.8. PCoA of interaction between developed cover and crop richness for pollinator community composition. All variables were treated as continuous to plot points, but colored and assigned shapes using categorical versions. Open points represent sites with low crop richness (3-14 unique crops) and closed points represent sites with high crop richness (>16 unique crops).

Comparison of farms and prairies

We observed differences in foliar insect and pollinator abundances between prairies and farms. Specifically, prairies supported fewer foliar pests and predators than urban and peri-urban farms (Table 1.4, Figs. 1.7a and 1.7b), whereas rural farms supported intermediate levels of these insects. On the other hand, prairies supported slightly more pollinators than urban and rural farms, although the differences were not significant (Table 1.4). Peri-urban farms supported an intermediate number of pollinators (Fig. 1.9c). The abundances of ground arthropods were similar between prairies and all farm types.

Table 1.4. Results of ANOVA and Tukey's HSD tests comparing arthropod abundance at prairies and farms. Values for Tukey's HSD are only provided if meaningful.

Guild	ANOVA Results	Tukey's HSD Prairie - Urban Farm	Tukey's HSD Prairie - Peri-Urban Farm	Tukey's HSD Prairie - Rural Farm
Ground Pests	p=0.10			
Ground Predators	p=0.08			
Foliar Pests	p=0.03*	p=0.05*	p=0.02*	
Foliar Predators	p<0.01*	p=0.02*	p=0.05*	
Pollinators	p=0.05*	p=0.07		p=0.06

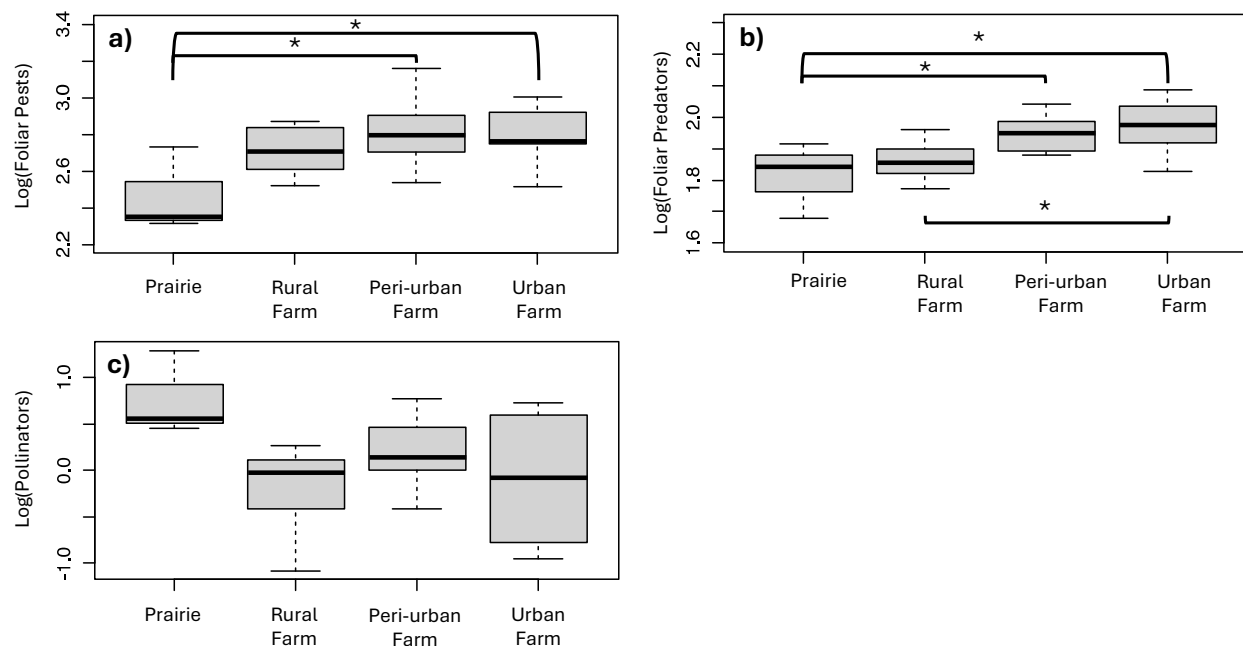


Figure 1.9. Boxplots comparing prairie and farm arthropod abundances for (a) foliar pests, (b) foliar predators, and (c) pollinators. Significant differences are denoted with an asterisk.

Community composition also differed between farms and prairies. Specifically, the

composition of ground predators and foliar pests at prairies varied significantly from all types of farms (for urban, peri-urban, and rural farms, respectively, ground predators $p=0.04$, $p<0.01$, $p<0.01$; foliar pests $p=0.02$, $p=0.03$, $p=0.04$; Table 1.5). Foliar predator communities on prairies were similar in composition to rural farms, but significantly different from urban and peri-urban farms ($p=0.05$, $p=0.02$).

Table 1.5. Pairwise PERMANOVA results of community composition comparing prairies to farms. Only significant p-values for pairwise tests are included (denoted by *).

Guild	Prairie - Urban Farm	Prairie - Peri-urban Farm	Prairie - Rural Farm
Ground Pests			
Ground Predators	$p<0.04^*$	$p=0.01^*$	$p<0.01^*$
Foliar Pests	$p=0.02^*$	$p=0.03^*$	$p=0.04^*$
Foliar Predators	$p=0.05^*$	$p=0.02^*$	
Pollinators			

Discussion

The purpose of this study was to understand how surrounding developed cover, farm size, and floral richness influence arthropod communities on vegetable farms, and to compare the habitat quality of these farms to restored prairies. We achieved this using a field study conducted over two years. Due to the broad scope and limited timeline of this project, arthropods were

grouped into coarse categories, with some identified to family and others to order. We expect that community compositions would have shown more differences between farm types, and between farms and prairies if all identifications were made to genus or species. However, we did still see patterns arise at this scale. Additionally, this study surveyed a small number of prairies (three). To our knowledge, however, this is one of the first studies to compare arthropod communities in vegetable farms and restored prairies.

Foraging strata vs feeding guilds

We expected local variables (farm size, crop richness) to primarily affect ground arthropods and our landscape variable (developed cover) to primarily affect foliar arthropods (including pollinators). Our findings did not support these hypotheses, as both ground and foliar arthropods responded to local and landscape variables. However, other studies have found that some arthropod groups may respond to a combination of local and landscape variables (Burks & Philpott, 2017; Maher et al., 2022; Philpott et al., 2014)

We also hypothesized that arthropod responses to local and landscape factors would vary depending on feeding guilds (pests, predators, or pollinators). Herbivores are reliant on the plant community (Agrawal et al., 2006), and therefore we expected pests to primarily respond to local variables such as cultivated farm size and crop richness. In contrast, we expected predators to respond primarily to landscape variables, as they may require additional food resources from the broader landscape if prey resources are scarce, and they are generally more mobile than pests (Karakoç et al., 2017; Langellotto & Denno, 2004). Pollinators are reliant on local floral and nesting resources, but also often have relatively large foraging ranges, and thus interact with the landscape (Kennedy et al., 2013). Therefore, we hypothesized that they would be influenced by a

combination of both local and landscape effects. In contrast, the results from our study suggest that pests responded to both local and landscape factors, but predators weren't consistently influenced by any variables. Pollinators did follow our predictions somewhat and were influenced by an interaction between local and landscape effects. Our findings suggest that unique combinations of foraging strata and feeding guilds create life histories that are sensitive to different local and landscape variables.

Landscape scale effects

The amount of developed cover (our landscape variable) surrounding the farms that we studied affected pests and predators. In particular, farms in developed/urban areas supported more ground pests and foliar predators. Isopods, the primary ground pest that we encountered, are known to be more prevalent in urban settings than rural ones (Hornung et al., 2007; Philpott et al., 2014; Vilisics et al., 2007). However, we did not expect a positive correlation between foliar predator abundances and developed cover. Our foliar predators were primarily parasitoid wasps, and previous work has found that their abundances generally decline with or do not correlate with increasing developed cover (Fenoglio & Salvo, 2011; Korányi et al., 2022). In our study, the positive correlation may be explained by the landscape-moderated concentration and dilution hypothesis (Burks & Philpott, 2017; Tschardt et al., 2012), where habitat loss in the urban landscape leads to increased concentration of arthropods in the remaining hospitable areas. If urban landscapes provide fewer resources or lower quality habitat (Raupp et al., 2010), then parasitoids and other foliar natural enemies may concentrate on urban farms because these sites have larger concentrations of prey and serve as refuge within a largely inhospitable urban landscape. Another possible explanation for the greater number of insect predators on urban

farms could be due to greater floral resources in these areas, as many predators and parasitoids rely on floral resources as adults (Burks & Philpott, 2017; Landis et al., 2000). To determine whether insect responses are indeed due to concentrated resource effects or greater floral resources within urban farms would require sampling arthropods in the surrounding landscape as well as quantifying floral abundance.

Local scale effects

The two local predictor variables (farm size and crop richness) affected insect abundances. We expected crop richness to decrease pest abundances by limiting the ability of pest insects to locate host plants and/or to increase predator abundances by providing alternative food resources. Increasing crop richness (especially via intercropping, or polyculture) is commonly used as a strategy on farms to decrease pest abundance (see the resource concentration hypothesis [RCH]; Root, 1973). In our study, crop richness increased ground pest abundances, which was contrary to our expectations. More recently, it has been discovered that the RCH typically does not hold for generalist herbivores and is not consistently true for specialist herbivores (Y. Chen et al., 2022). Isopods, the most common ground pests in our study, are opportunistic herbivores and consume a wide variety of fruits and vegetables (Souty-Grosset & Faberi, 2018). Thus, it is unsurprising that they do not conform to the RCH. Instead, having a variety of different crop types might directly increase these pest numbers by providing alternative resources for the opportunistic feeders and increasing heterogeneity in leaf litter, improving abiotic factors such as soil moisture .

For foliar pests and predators, there was an interaction between crop richness and farm size, suggesting that both are important for affecting foliar insect numbers. Specifically,

increasing crop richness reduced foliar pest abundance on small farms only and increased foliar predator abundances only on large farms. Likewise, altering crop richness affected the composition of the foliar pest community but in different ways depending on farm size. Large farms had similar pest communities while small farms had dissimilar pest communities. Based on the literature, we expected these respective relationships (greater crop richness to reduce pest abundance and increase predator abundance) but had not anticipated them to be mediated by farm size. These responses could arise if foliar arthropods are responding to the densities of each crop type rather than crop richness and farm sizes per se. For example, farms with high crop richness may only have a few plants of each crop type. On small farms, plant numbers are even lower, which could reduce pest numbers as they are unable to easily exploit any one type of crop or find host plants. In contrast, large farms with high crop richness could comparatively have more of each crop type because they have more planting space. Therefore, larger farms would offer predators a variety of different food types in higher densities. These results support previous work showing the benefits of increasing crop richness for pest suppression on farms but highlight that the mechanisms could vary by farm size (MacArthur & Wilson, 1967; Philpott et al., 2023). These responses suggest that both farm size and heterogeneity (i.e., crop richness) are needed to affect foliar pest and predator abundances on urban farms, and that there may be a critical mass required for crop species to be effective.

Interactions between landscape and local variables

Landscape moderation theory suggests that local management practices, such as plant richness, vary with landscape complexity (Tschamntke et al., 2012), and we observed this with the foliar pest and pollinator communities. In particular, developed cover and crop richness

interacted to affect foliar pest community composition, where increasing crop richness altered foliar pest composition but only on rural farms. In our study, rural farm sites were surrounded by corn and soy fields, with few natural areas. These surrounding agricultural fields may be source habitats for foliar pests such as thrips and whiteflies, and spill over onto the rural farms in response to changes in crop richness. In contrast, peri-urban and urban farms in our study were incorporated into much more complex and diverse landscapes, with parks, lawns, and landscaping, which may not be ideal habitats for crop pests. Therefore, altering crop richness on urban and peri-urban farms may not alter pest community composition because there are fewer pests in the surrounding landscape to spill over into these farms. Thus, managing pests through changes in crop richness may depend on landscape context, specifically whether there is an adequate source habitat for pests nearby.

Pollinators were also influenced by this interaction between developed cover and crop richness. For this group, we saw that differences in crop richness only affected pollinator abundance in urban systems, where greater crop richness resulted in fewer pollinators. Typically, greater plant richness fosters higher abundances of pollinators (Neumann et al., 2024). We suspect that the pattern we observed in pollinator abundance was tied to floral abundance (Plascencia & Philpott, 2017). Urban farms tend to be smaller in size, and if they plant a diverse array of crops, there are fewer individual plants of each type. At peri-urban and rural farms, this pattern was likely not as pronounced because there were sufficient quantities of each type of crop plant. Frequent soil disturbance at the urban farms may also be playing a role, as soil is a nesting habitat for many bees (Antoine et al., 2025; Ullman et al., 2020). Urban farms are often limited in production space and therefore utilize succession planting, crop rotation, inter-planting, and multiple cropping cycles during a growing season (Arnold, 2022; Olivier, 2018). This intensive

management may be leading to more frequent soil disturbances in the production area, affecting arthropods (Matteson & Langellotto, 2010). Early-season crops are typically removed before they flower, so increasing crop richness in this case also did not necessarily indicate an increase in floral resources.

Comparison of prairies and farms

To understand the value of vegetable farms (and specifically UA) as habitat for arthropods, we need to determine how they compare to natural areas, which are typically considered to be their native environments. To our understanding, this is one of the only studies that compares arthropod communities at restored prairies to those of farms across an urbanization gradient. Therefore, despite limitations of sampling relatively few restored prairies in only one year, we find value in the novelty of this study, and believe it has the capacity to serve as a baseline for future research.

We found that prairies had significantly fewer foliar arthropods (pests and predators) than urban and peri-urban farms. Many herbivorous pests (especially those prone to outbreaks like aphids, thrips, and whiteflies) are associated with crops and ornamental plants commonly grown on farms (Messelink et al., 2021; Saurabh et al., 2021). We also observed greater foliar predator abundances at farms compared to prairies. Because most of the predators observed in our study were specialist parasitic wasps, their responses were likely the result of increases in foliar pests on urban and peri-urban farms.

Pollinator abundances were greater at prairies compared to both urban and rural farms, though the pairwise comparisons were only marginally significant. Pollinator compositions were not significantly different between farms and prairies. This suggests that farms and prairies may

function as habitats of similar quality for pollinators, as they can support similar quantities and types of these beneficial insects. Thus, our data suggest that farms have a high potential to harbor biodiversity and support beneficial insects, which provide vital ecosystem services. Ground pest abundances and compositions also did not differ between farms and prairies. The ground pests that we captured were generalists and not expected to be sensitive to the plant community (Y. Chen et al., 2022).

Management implications and future directions

In our study, we measured the effects of developed land cover, crop richness, and farm size on insect communities. While the amount of variation explained in our models was relatively high for foliar insects ($R^2=0.48-0.64$), for pollinators and ground insects, the amounts of variation explained were relatively low ($R^2=0.09-0.34$). Other unmeasured variables might be more important in shaping the abundance and composition of ground insects and pollinators. Factors like soil management practices (tilling, mulching, etc.) and soil conditions are very influential to the ground-dwelling arthropods (Moss & Hassall, 2006; Thorbek & Bilde, 2004). Floral abundance and total floral richness (beyond crop richness) may also explain more variation in pollinator communities (Cohen et al., 2021; Philpott et al., 2023; Plascencia & Philpott, 2017). Additionally, habitat connectivity is likely affecting arthropods in these systems as well and should be considered in future work.

In several cases, we found that crop richness was a predictor of arthropod abundance and composition, suggesting that arthropod management could be achieved by altering levels of crop richness. However, the efficacy of crop richness to alter insect communities was context dependent. It will be important to consider the size and location of the farm within the landscape.

We compared insect communities on farms to those in nearby prairies, which allowed us to understand their abilities to support diverse arthropods. This information could inform conservation and city planning efforts through the lens of habitat quality. The ability of other forms of green space (i.e., parks, cemeteries, vacant lots) to support biodiversity should also be compared to that of farms. With the percentage of the global population living in cities expected to increase from 45% in 2025 to 66% by 2050 (United Nations, 2025), urban sprawl and intensification are inevitable. Understanding how different forms of green spaces support insects is thus needed to mitigate insect biodiversity decline in the face of these changes.

Conclusion

We found that ground and foliar arthropods both responded to local and landscape variables. Crop richness, a local variable, was important in shaping the abundances and compositions of pests, predators, and pollinators. However, its effectiveness was often context dependent.

To our understanding, this is one of the first studies of its kind to be conducted in the Central United States, and specifically in Kansas City. Each city has a unique structure, varying in things like development density and amount of green space. Cities also may exist in different climates, which may influence patterns in arthropod communities as well. Therefore, patterns observed in one city may not be entirely comparable to another city. Our study provides value in being one of the first to represent this area of the United States. This study was also novel in its comparison of prairies to vegetable farms across an urbanization gradient.

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Chapter 2 - Effects of habitat type, floral richness, and site size on pollinator community structure and foraging behavior

Abstract

Over one-third of global food production relies on arthropod- or animal-mediated pollination. However, many pollinating arthropods are in decline. Urbanization is one of the contributing factors that has been tied to these declines. Green spaces in cities have been recognized for their ability to alleviate some of the effects of urbanization on arthropods and other wildlife. One particular form of green space, urban agriculture (UA), has been identified as a potential “hotspot” for pollinators. UA includes spaces like backyard or community gardens and market farms. It is unclear, however, how UA compares to natural areas like restored prairies as pollinator habitat. Additionally, few studies examine whether functional behavior of pollinators (i.e., specialization, modularity) may differ between habitat types. In this study, we compared pollinator abundance, richness, species composition, and functional behavior between vegetable farms and restored prairies. Floral richness and site size were also examined as potential underlying factors driving the structure and function of pollinator networks. We found that pollinator species composition differed between rural farms and prairies as well as across urban and rural farms. However, no significant differences in pollinator abundance, richness, modularity, or specialization were observed across habitat types. Additionally, we found that floral richness influenced the functional behavior of pollinator networks. Our findings suggest that some vegetable farms may function in ecologically similar ways as prairies, but that they may not support the same pollinator species. Research should continue to explore other underlying mechanisms shaping the structure and functional behavior of pollinator communities.

Introduction

Cities around the world are expanding, and natural areas are shrinking (Ren et al., 2023). As a result, global declines in biodiversity are being observed across numerous taxa, but especially arthropods (Sánchez-Bayo & Wyckhuys, 2019). To mitigate habitat loss and promote biodiversity, urban green spaces have been introduced in cities (Hall et al., 2017). Urban green spaces come in a wide variety of forms, including, but not limited to, parks, golf courses, trails, gardens, forests, and green roofs. Urban farms and gardens are both forms of green space that have gained popularity in recent years and have been suggested to be important for harboring biodiversity in cities (Di Pietro et al., 2018; Gaston & Gaston, 2010). Many vegetable farms and gardens also grow exotic and/or native flowers for a variety of purposes. Some urban farmers grow cut flowers as an additional income source (Curtis & Stock, 2023), while others hope to draw in pollinators and other beneficial insects to reap extra ecosystem services for their primary crops (Haaland et al., 2011). Regardless of the intention of farmers in planting flowers, arthropods that rely on floral resources can benefit from them.

The prevalence of floral resources has led urban gardens to be recognized as beneficial habitat patches for pollinators, but the interpretations of the conservation value of this type of habitat are conflicting. Across a variety of types of green spaces, the effects of urbanization on pollinators remain unclear. An array of findings have been published; conclusions range from urbanization negatively affecting pollinators to having neutral, or even positive, effects in several different metrics (Silva et al., 2021; Wenzel et al., 2020). For example, some studies find that pollination success and pollinator visitation are relatively unaffected in urban centers (Lowenstein et al., 2014; Verboven et al., 2012), while others find lower pollination success and

fewer visits by pollinators to flowers in urban habitats (Andrieu et al., 2009; Pellissier et al., 2013).

Within farms and gardens specifically, few studies have been done that consider the influences of urbanization on pollinator communities (Silva et al., 2021). More research needs to be done to determine if urban landscapes are the best places to concentrate conservation efforts and establish habitat patches for pollinators.

While adding urban farms and gardens to a city may augment pollinators, it is unclear how these green spaces compare to other forms of habitat (especially natural areas) in their ability to harbor diverse assemblages of pollinators. Home gardens have been shown to support different pollinator assemblages than various types of natural areas, such as forests and bushlands (Martins et al., 2017; Prendergast & Ollerton, 2021; Youngsteadt & Keighron, 2023). Therefore, these habitats may not be equal in conservation value. Gardens may only be supporting common pollinator species like honey bees and bumble bees, rather than supporting species of conservation concern, which may require natural habitats. Different types of urban agriculture (UA) may also support different pollinator assemblages and thus have different conservation value. Though similar in function, community gardens, institutional gardens, and market farms are often subject to different management strategies than home gardens, and thus likely differ in their habitat quality. Many studies examine home gardens as pollinator habitat, but studies examining market farms and institutional gardens are much less common.

In addition to understanding how the pollinator community structure varies on urban farms, understanding the functional behavior of these communities is important as well. One way to examine functional behavior is through plant-pollinator bipartite networks, which have proven useful for informing conservation management practices (Elle et al., 2012). Bipartite networks

are often used in ecology to depict the interactions between organisms of two trophic levels. This type of network is made up of two rows of species. The top row is a list of species from the higher trophic level (i.e., pollinators), and the bottom row is a list of species from the lower trophic level (i.e., plants). Between these two rows of species are lines or bars that depict the interactions observed between trophic levels (i.e., pollinators visiting plants). This type of network does not directly show the competition that may occur between two organisms within the same trophic level. Therefore, we cannot directly measure the competition effects that two pollinators may have on one another, but we can infer competition if two pollinators are both found visiting the same species of flower.

A network approach can help identify which plants and how many different types of plants pollinators visit (i.e., specialization levels), and this information can then be used to inform habitat planting mixes (Elle et al., 2012). Pollinator community structure can also be assessed using a network approach. For example, modularity, a common network metric, describes the structural organization of communities and can act as a measure of network robustness (Newman, 2006). Bipartite networks can be quantified in urban systems and compared to natural areas. These comparisons allow us to determine the suitability of various urban green spaces for pollinator conservation.

To understand the structure and function of pollinator communities on urban farms, the driving questions for this study are as follows:

- (1) How does pollination community structure (i.e., pollinator abundance, species richness, and community composition) change on farms across a gradient of urbanization, and how do these metrics compare between farms and prairies?

- (2) How does the functional behavior of pollinator communities (i.e., specialization, modularity) on farms vary across a gradient of urbanization, and between farms and prairies?
- (3) Do local variables like floral richness and site size affect pollination community structure and function?

We hypothesize that pollinators will be more abundant on urban farms than rural ones, but that prairies will have more pollinators than any type of farm because pollinators tend to prefer native plants (Chrobock et al., 2013; Morales & Traveset, 2009). We also expect that pollinator richness will be greater on rural farms than on urban farms, but that prairies will have the greatest species richness. We hypothesize that pollinator community composition will differ between urban and rural farms, and that peri-urban farms will be similar to both urban and rural farms, as several bee species are known to be better adapted to urban settings (Hernandez et al., 2009). We predict that rural farms will have the most similar community composition to prairies because natural areas and gardens tend to support different assemblages of pollinator species (Prendergast et al., 2022), but rural landscapes contain more natural area compared to urban landscapes. Thus, the source populations of pollinators available to colonize and forage at rural gardens should be more similar to those in natural areas.

We hypothesize that network modularity will be greater on rural farms than urban ones, and that prairies will be more modular than all types of farms. To examine foraging behavior, we will use network specialization (H_2') and species specialization (d') as our proxies. We hypothesize that rural farms will have overall more specialized networks (greater H_2') than urban farms, and that many species will show greater specialization (d') at rural farms than urban farms because urban areas tend to better support generalist species (Buchholz & Egerer, 2020; Martins et al.,

2017). We also predict that prairie pollinator networks will be more specialized than any type of farm because native plants tend to support more specialized pollinator interactions (Seitz et al., 2020).

In regards to floral richness and site size, we anticipate that pollinator abundance and species richness will positively correlate with these factors. We also predict that community composition will differ between sites of high and low floral richness, and between larger and smaller sites. We predict that modularity will be greater at larger sites and sites with high floral richness. Finally, we expect foraging behavior to be more specialized at sites of high floral richness and larger size.

Methods

Twelve agricultural sites, in the form of a combination of market farms, school farms, community gardens, and church gardens, were visited in 2025 as study sites (Fig. 2.1). Many of these sites were not certified organic; however, all reported using organic growing techniques. These sites will be collectively referred to as “farms” for the purpose of this report. Each farm grew a variety of fruits, vegetables, herbs, and ornamental flowers. Three restored prairie sites were selected in this area along a gradient of size and urbanization. Pollinators visiting vegetables, herbs, ornamental flowers, and wildflowers at each site were sampled using hand-held insect vacuums.

Site selection

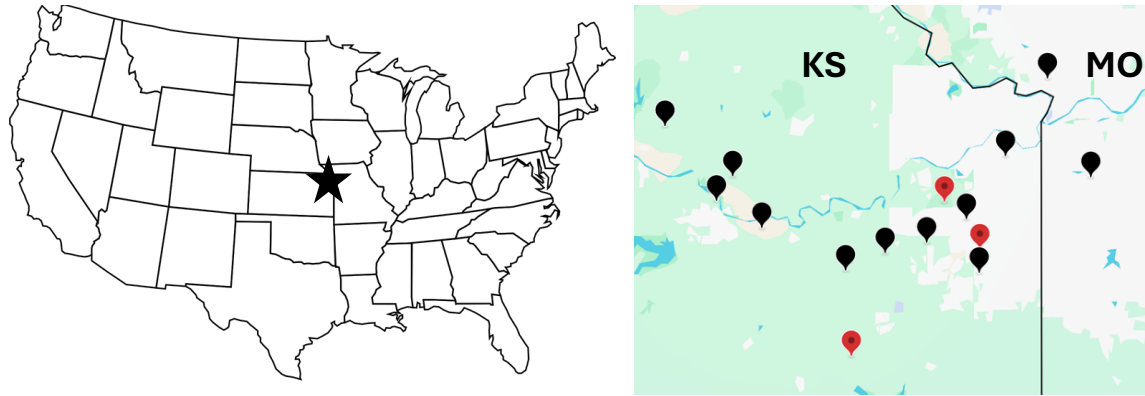


Figure 2.1. Map of study sites within the Kansas City metropolitan area. Black pins represent farm sites and red pins represent prairie sites.

Farms were generally located within the Kansas City metropolitan region, which extends through Northeastern Kansas and Northwestern Missouri (Fig 2.1). Different management strategies and production intensity were practiced depending on the size of the property and the end-market for the products (e.g., sold vs donated). These farms were selected along a gradient of urbanization, ranging from urban to rural. ArcGIS Pro and the USDA NASS Cropland Data Layer (USDA NASS, 2024) were used to determine this gradient. Around each site, a 1 km buffer was generated, and the degree of urbanization was determined by calculating the percentage of ‘developed’ land class within the buffer. This buffer size was selected as an average foraging distance for pollinators (Gathmann & Tschardtke, 2002; Greenleaf et al., 2007). ‘Developed’ land was considered the sum of the high, medium, low, and open space developed classes from established Cropland Data Layer categories. Sites were also chosen to encompass a range of sizes. Farm size was measured using Google Earth by drawing polygons around the cultivated portion of the site and calculating the area. Monthly surveys of the plant community were also conducted at each site by recording ornamental, vegetable, herb, and weedy plants that were in observed flower to genus or species.

The three prairie sites were selected using the same techniques previously described to choose farms (Fig. 2.1). All prairies were restored at least five years prior to this study and were seeded with mixes of forbs and grasses. During the study, these sites were not mowed, burned, or sprayed.

Pollinator collection

Five transects were established per site in vegetable crops, wildflowers, and/or ornamental flowers. These transects were spaced out to encompass the full area of the farm, and were selected to help quantify the pollinators visiting the whole farm, as opposed to only targeting showy flowering plants. Ten-minute timed observations were conducted at each transect once per month for a duration of three months (June, July, August) in 2025. The plant that each insect was collected or observed on was identified in the field to genus and/or species and recorded.

Pollinating insects were sampled via 10-minute timed observations walking back and forth along transects. Insects that were interacting with floral reproductive parts were collected from flowers using hand-held insect vacuums (modified cordless Skil hand vacuums). Some pollinators of conservation concern were not captured and were instead identified and recorded as either family or genus. These included groups like bumble bees.

Insects that were captured by vacuum were then transferred into 120 ml plastic sample cups and stored in a -20° C freezer until the time of processing. Bees and syrphid flies were pinned and identified to species using taxonomic keys and a microscope. Bees were identified and then verified by Joel Gardner at Washington State University. A variety of taxonomic keys and resources were used to determine identifications (Arduser, 2019; *Discover Life*, 2025;

Gardner & Gibbs, 2022; Portman et al., 2022). Syrphid flies were also identified to species using various resources (Dankowicz & Dankowicz, 2025; Miranda et al., 2013; Skevington et al., 2019).

Statistical analyses

All sampling sessions across the season were pooled by site. Differences in abundance and species richness across habitat type (prairie, urban farm, peri-urban farm, rural farm) were analyzed using ANOVA and Tukey's HSD. Species richness was standardized using rarefaction. We also compared the number of species shared between different types of farms and between farms and prairies using a Venn diagram from the packages *VennDiagram* and *grid* (H. Chen & Boutros, 2011; R Core Team, 2024). Community composition was compared for both the plant and pollinator data we collected from our observed interactions. To do this, we used pairwise PERMANOVA testing (*pairwiseAdonis* package in R; Martinez Arbizu, 2017) and principal coordinate analyses (PCoA) for visualization. PCoA plots and PERMANOVAs were based on a Jaccard distance matrix and 999 permutations.

To examine functional behavior, we quantified modularity and specialization (network and species level). Modularity measures the degree to which distinct groupings of interactions (modules) occur within a community. High modularity indicates that the community consists of modules that, within themselves, are highly connected and have little connectivity with other modules. Ecologically, communities that are more modular are more resistant to disturbance (Stouffer & Bascompte, 2011). The functions 'walktrap.community', 'fastgreedy.community' from the *igraph* package were used to calculate modularity (Csárdi & Nepusz, 2006).

Specialization can be measured at both the species level and network level. Species-level

specialization (d') is a measure of the dietary specialization for a single species; we calculated this for a select group of pollinators. This metric is measured on a scale of zero to one, where zero indicates the pollinator species interacts with all potential plant species (a generalist diet), and a score of one indicates that the pollinator species interacts with only one potential plant species (a specialist diet). We calculated d' using the function 'dfun' from the *bipartite* package in R (Dormann et al., 2009). Network level specialization (H_2') provides a summary of the specialization of all species within a network (Blüthgen et al., 2006). H_2' is also measured on a scale of zero to one, where zero represents an entirely generalized network (i.e., complete overlap in floral visitation among visitors), and one represents an entirely specialized network (i.e., no overlap in floral visitation). The function 'H2fun' from the *bipartite* package was used to calculate H_2' (Dormann et al., 2009). Bipartite networks were constructed for each site using R statistical software and the packages *bipartite*, *igraph*, and *vegan* (Csárdi & Nepusz, 2006; Dormann et al., 2009; Oksanen et al., 2025). A collective regional network was also constructed by compiling the plant-pollinator interaction data from all sites.

We used null model analyses to determine whether observed modularity and specialization values deviated from random networks. Z-scores were calculated from the mean and standard deviation of null model runs to compare metrics across sites and were based on 1,000 replicates. Comparisons were performed using either ANOVA and Tukey's HSD, or linear models, depending on whether the predictor variable was categorical or continuous.

Truly random null models allow all species of one trophic level to interact with all other species of the other trophic level. However, these null models do not reflect ecological reality as pollinators often exhibit dietary preferences, and may have morphological restraints (e.g., body size, tongue length) which inhibit their ability to feed on and pollinate certain plants. To account

for this, we used a null model designed by Spiesman and Gratton (2016), which preserves the number of plant and pollinator species, the number of interactions, and connectance of the network. This null model also bans ‘forbidden links’ so that only interactions that are known to occur between species are permitted. This is done by comparing local networks to a regional network. The regional network was created by pooling all local interactions together into one large meta network, which depicted all potential interactions that occurred between plants and pollinators in our study region. Because we pooled interaction data by site across the whole season, we did not include forbidden links based on phenological mismatches between plants and pollinators. All statistical analyses were performed in R statistical software (version 4.4.2, R Core Team, 2024).

Results

We observed a total of 2,412 bees and syrphid flies during this study. Specimens were only included in the final analyses if they could be identified to species or species group. Most specimens were bees, which constituted 2,229 individuals belonging to 69 species (or species groups) from five families. The remaining 183 individuals represented 17 species of the family Syrphidae. Interactions were recorded on 66 flowering plant species. Complete lists of insect and plant species can be found in Tables 2.1 and 2.2, respectively.

Table 2.1. Alphabetized lists of bee and syrphid species observed during this study. Total abundances are listed in parentheses after each species name.

Bee species	Total count	Bee species (cont'd)	Total count	Syrphid species	Total count
<i>Agapostemon sericeus</i>	(2)	<i>Lasioglossum imitatum</i>	(195)	<i>Allograpta obliqua</i>	(5)
<i>Agapostemon texanus</i>	(1)	<i>Lasioglossum lineatulum</i>	(1)	<i>Epistrophe metcalfi</i>	(1)
<i>Agapostemon virescens</i>	(12)	<i>Lasioglossum oceanicum</i>	(1)	<i>Eristalis arbustorum</i>	(32)
<i>Anthidiellum notatum</i>	(1)	<i>Lasioglossum smilacinae</i>	(1)	<i>Eristalis flavipes</i>	(1)
<i>Anthidium manicatum</i>	(1)	<i>Lasioglossum subviridatum</i>	(1)	<i>Eristalis stipator</i>	(25)
<i>Anthidium oblongatum</i>	(9)	<i>Lasioglossum trigeminum</i>	(22)	<i>Eristalis transversa</i>	(10)
<i>Apis mellifera</i>	(667)	<i>Lasioglossum versatum</i>	(20)	<i>Helophilus latifrons</i>	(1)
<i>Augochlora pura</i>	(8)	<i>Lasioglossum weemsi</i>	(4)	<i>Meromacrus acutus</i>	(1)
<i>Augochlorella aurata</i>	(92)	<i>Lasioglossum zephyrus</i>	(2)	<i>Orhonevra nitida</i>	(1)
<i>Augochlorella persimilis</i>	(20)	<i>Megachile brevis</i>	(2)	<i>Palpada vinetorum</i>	(12)
<i>Augochloropsis metallica</i>	(2)	<i>Megachile campanulae</i>	(2)	<i>Paragus haemorrhous</i>	(24)
<i>Augochloropsis viridula</i>	(1)	<i>Megachile concinna</i>	(1)	<i>Pseudodoros clavatus</i>	(3)
<i>Bombus spp.</i>	(608)	<i>Megachile exilis</i>	(10)	<i>Sphaerophoria contigua</i>	(4)
<i>Calliopsis andreniformis</i>	(10)	<i>Megachile inimica</i>	(1)	<i>Syrpita pipiens</i>	(7)
<i>Ceratina mikmaqi</i>	(116)	<i>Megachile mendica</i>	(7)	<i>Syrphus ribesii</i>	(1)
<i>Ceratina strenua</i>	(24)	<i>Megachile petulans</i>	(3)	<i>Toxomerus geminatus</i>	(5)
<i>Coelioxys germanus</i>	(2)	<i>Megachile rotundata</i>	(9)	<i>Toxomerus marginatus</i>	(50)
<i>Coelioxys sayi</i>	(3)	<i>Megachile texana</i>	(3)		
<i>Coelioxys texanus</i>	(1)	<i>Melissodes bimaculatus</i>	(42)		
<i>Halictus confusus</i>	(19)	<i>Melissodes comptoides</i>	(6)		
<i>Halictus ligatus</i>	(70)	<i>Melissodes coreopsis</i>	(1)		
<i>Halictus parallelus</i>	(1)	<i>Melissodes denticulatus</i>	(12)		
<i>Halictus rubicundus</i>	(9)	<i>Melissodes despondus</i>	(3)		
<i>Heriades carinata</i>	(11)	<i>Melissodes trinodis</i>	(5)		
<i>Heriades leavitti</i>	(9)	<i>Peponapis pruinosa</i>	(15)		
<i>Heriades variolosa</i>	(7)	<i>Protandrena albitarsis</i>	(1)		
<i>Hylaeus leptcephalus</i>	(1)	<i>Ptilothrix bombiformis</i>	(2)		
<i>Hylaeus mesillae (complex)</i>	(13)	<i>Sphecodes cf. atlantis</i>	(1)		
<i>Hylaeus saniculae</i>	(1)	<i>Sphecodes cf. prosphorus</i>	(1)		
<i>Hylaeus sp. 1</i>	(11)	<i>Stelis louisae</i>	(3)		
<i>Hylaeus sp. 2</i>	(3)	<i>Epimelissodes obliquus</i>	(21)		
<i>Lasioglossum bruneri</i>	(1)	<i>Triepeolus concavus</i>	(4)		
<i>Lasioglossum disparile</i>	(5)	<i>Triepeolus lunatus</i>	(6)		
<i>Lasioglossum ellisiae</i>	(6)	<i>Xylocopa virginica</i>	(18)		
<i>Lasioglossum hitchensi</i>	(56)				

Table 2.2. Alphabetized list of plant species and their respective numbers of recorded interactions. Note that this is not an exhaustive list of all plants present at each site; it is only representative of plants that pollinators were observed visiting during our study.

Plant species	Number of recorded interactions	Plant species (cont'd)	Number of recorded interactions
<i>Achillea millefolium</i>	(7)	<i>Inula helenium</i>	(1)
<i>Agastache foeniculum</i>	(5)	<i>Lactuca serriola</i>	(3)
<i>Alcea</i> spp.	(10)	<i>Lagerstroemia</i> spp.	(12)
<i>Allium</i> spp.	(1)	<i>Lavendula</i> spp.	(14)
<i>Anethum graveolens</i>	(29)	<i>Leucanthemum vulgare</i>	(4)
<i>Artemisia</i> spp.	(14)	<i>Liatris</i> spp.	(39)
<i>Asclepias tuberosa</i>	(18)	<i>Limonium sinuatum</i>	(2)
<i>Asclepias verticillata</i>	(2)	<i>Melilotus albus</i>	(2)
<i>Brassica</i> spp.	(9)	<i>Mentha</i> spp.	(202)
<i>Calendula officinalis</i>	(4)	<i>Monarda</i> spp.	(213)
<i>Capsicum</i> spp.	(4)	<i>Ocimum basilicum</i>	(78)
<i>Celosia</i> spp.	(11)	<i>Origanum vulgare</i>	(93)
<i>Centaurea cyanus</i>	(42)	<i>Phlox</i> spp.	(7)
<i>Chamaecrista fasciculata</i>	(8)	<i>Physostegia virginiana</i>	(3)
<i>Convolvulus arvensis</i>	(9)	<i>Plantago</i> spp.	(12)
<i>Coreopsis</i> spp.	(51)	<i>Pycnanthemum muticum</i>	(203)
<i>Cosmos bipinnatus</i>	(10)	<i>Pycnanthemum tenuifolium</i>	(10)
<i>Cucumis</i> spp.	(116)	<i>Ratibida</i> spp.	(85)
<i>Cucurbita pepo</i>	(21)	<i>Rudbeckia</i> spp.	(83)
<i>Dalea purpurea</i>	(13)	<i>Silphium integrifolium</i>	(425)
<i>Daucus carota</i>	(39)	<i>Solanum lycopersicum</i>	(3)
<i>Echinacea</i> spp.	(59)	<i>Solanum rostratum</i>	(2)
<i>Erigeron</i> spp.	(7)	<i>Stachys officinalis</i>	(11)
<i>Eryngium yuccifolium</i>	(62)	<i>Tagetes</i> spp.	(3)
<i>Eschscholzia</i> spp.	(28)	<i>Tanacetum parthenium</i>	(1)
<i>Gaillardia pulchella</i>	(108)	<i>Torilis arvensis</i>	(8)
<i>Gnaphalidae</i>	(12)	<i>Trifolium</i> spp.	(42)
<i>Gomphrena globosa</i>	(11)	<i>Verbascum</i> spp.	(1)
<i>Helianthus mollis</i>	(1)	<i>Verbena bonariensis</i>	(12)
<i>Helianthus</i> spp.	(10)	<i>Verbena</i> spp.	(3)
<i>Heliopsis helianthoides</i>	(8)	<i>Vernonia</i> spp.	(6)
<i>Hibiscus</i> spp.	(3)	<i>Vicia</i> spp.	(2)
<i>Hylotelephium</i> spp.	(14)	<i>Zinnia elegans</i>	(81)

Nine pollinator species were unique to prairies, and 45 species were unique to farms collectively (urban, peri-urban, and rural combined); 32 species were found at both prairies and some type of farm (Fig. 2.2a). Prairies and every type of farm shared 16 pollinator species. Table 2.3 lists all pollinator species by the habitat type in which they were found. Ten plant species were unique to prairies, and 39 species were unique to farms collectively; eight species were found at both prairies and farms (Fig. 2.2b). Only one plant species (*Daucus carota*) was common between prairies and every farm type. Table 2.4 lists all plant species by the habitat type in which they were found.

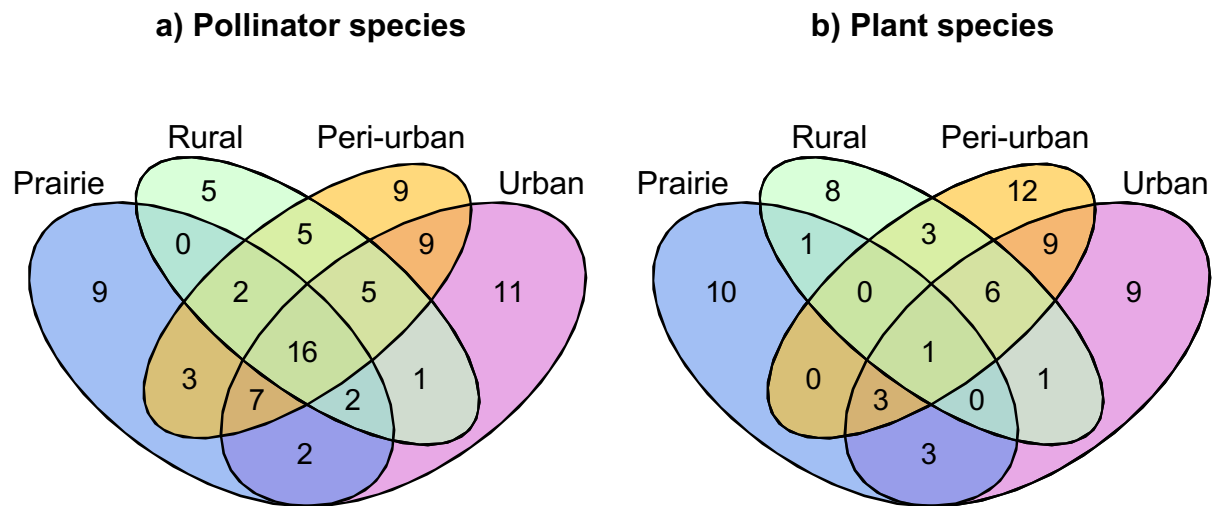


Figure 2.2. Venn diagrams of the number of shared (a) pollinator species and (b) plant species by habitat type: prairie, rural farm, peri-urban farm, and urban farm.

Table 2.3. List of pollinator species (bees and syrphid flies) by habitat type(s) they were observed in.

Prairies only	Prairies and peri-urban farms	Prairies and peri-urban/urban farms
<i>Agapostemon texanus</i>	<i>Agapostemon virescens</i>	<i>Ceratina strenua</i>
<i>Anthidiellum notatum</i>	<i>Lasioglossum versatum</i>	<i>Eristalis arbustorum</i>
<i>Augochloropsis viridula</i>	<i>Melissodes despondus</i>	<i>Halictus confusus</i>
<i>Eristalis flavipes</i>		<i>Heriades carinata</i>
<i>Lasioglossum smilacinae</i>	Prairies and urban farms	<i>Heriades leavitti</i>
<i>Meromacrus acutus</i>	<i>Coelioxys sayi</i>	<i>Heriades variolosa</i>
<i>Orthonevra nitida</i>	<i>Halictus rubicundus</i>	<i>Xylocopa virginica</i>
<i>Protandrena albitarsis</i>		
<i>Ptilothrix bombiformis</i>	Rural/peri-urban farms	Rural/peri-urban/urban farms
	<i>Agapostemon sericeus</i>	<i>Calliopsis andreniformis</i>
Rural farms only	<i>Lasioglossum ellisiae</i>	<i>Lasioglossum trigeminum</i>
<i>Helophilus latifrons</i>	<i>Lasioglossum zephyrus</i>	<i>Melissodes bimaculatus</i>
<i>Lasioglossum bruneri</i>	<i>Megachile brevis</i>	<i>Epimelissodes obliquus</i>
<i>Lasioglossum disparile</i>	<i>Triepeolus concavus</i>	<i>Syritta pipiens</i>
<i>Lasioglossum oceanicum</i>		
<i>Syrphus ribesii</i>	Rural/urban farms	
	<i>Peponapis pruinosa</i>	Prairies and rural/peri-urban/urban farms
Peri-urban farms only	Peri-urban/urban farms	<i>Apis mellifera</i>
<i>Allograpta obliqua</i>	<i>Augochloropsis metallica</i>	<i>Augochlora pura</i>
<i>Anthidium manicatum</i>	<i>Coelioxys germanus</i>	<i>Augochlorella aurata</i>
<i>Epistrophe metcalfi</i>	<i>Hylaeus sp. 2</i>	<i>Augochlorella persimilis</i>
<i>Halictus parallelus</i>	<i>Lasioglossum weemsi</i>	<i>Bombus spp.</i>
<i>Lasioglossum lineatulum</i>	<i>Megachile campanulae</i>	<i>Ceratina mikmaqi</i>
<i>Megachile inimica</i>	<i>Megachile exilis</i>	<i>Eristalis transversa</i>
<i>Megachile texana</i>	<i>Megachile petulans</i>	<i>Halictus ligatus</i>
<i>Toxomerus geminatus</i>	<i>Melissodes denticulatus</i>	<i>Hylaeus mesillae agg.</i>
<i>Triepeolus lunatus</i>	<i>Pseudodoros clavatus</i>	<i>Lasioglossum hitchensi</i>
		<i>Lasioglossum imitatum</i>
Urban farms only	Prairies and rural/peri-urban farms	<i>Megachile mendica</i>
<i>Anthidium oblongatum</i>		<i>Melissodes trinodis</i>
<i>Coelioxys texanus</i>	<i>Eristalis stipator</i>	<i>Palpada vinetorum</i>
<i>Hylaeus leptocephalus</i>	<i>Melissodes comptoides</i>	<i>Paragus haemorrhous</i>
<i>Hylaeus saniculae</i>		<i>Toxomerus marginatus</i>
<i>Lasioglossum subviridatum</i>	Prairies and rural/urban farms	
<i>Megachile concinna</i>		
<i>Megachile rotundata</i>	<i>Hylaeus sp. 1</i>	
<i>Melissodes coreopsis</i>	<i>Sphaerophoria contigua</i>	
<i>Sphecodes cf. atlantis</i>		
<i>Sphecodes cf. prosphorus</i>		
<i>Stelis louisae</i>		

Table 2.4. List of plant species by habitat type(s) they were observed in.

Prairies only	Urban farms only	Prairies and peri-urban/urban farms
<i>Agastache foeniculum</i>	<i>Allium spp.</i>	<i>Coreopsis spp.</i>
<i>Asclepias verticillata</i>	<i>Artemisia spp.</i>	<i>Monarda spp.</i>
<i>Chamaecrista fasciculata</i>	<i>Capsicum spp.</i>	<i>Ratibida spp.</i>
<i>Dalea purpurea</i>	<i>Hylotelephium spp.</i>	
<i>Erigeron spp.</i>	<i>Physostegia virginiana</i>	Rural/peri-urban/urban farms
<i>Eryngium yuccifolium</i>	<i>Solanum lycopersicum</i>	<i>Cucumis spp.</i>
<i>Helianthus mollis</i>	<i>Solanum rostratum</i>	<i>Echinacea spp.</i>
<i>Heliopsis helianthoides</i>	<i>Verbena spp.</i>	<i>Mentha spp.</i>
<i>Liatris spp.</i>	<i>Vernonia spp.</i>	<i>Ocimum basilicum</i>
<i>Pycnanthemum tenuifolium</i>		<i>Rudbeckia spp.</i>
	Prairies and rural farms	<i>Zinnia elegans</i>
Rural farms only	<i>Melilotus albus</i>	
<i>Brassica spp.</i>		Prairies and rural/peri-urban/urban farms
<i>Cucurbita pepo</i>	Prairies and urban farms	<i>Daucus carota</i>
<i>Eschscholzia spp.</i>	<i>Asclepias tuberosa</i>	
<i>Inula helenium</i>	<i>Silphium integrifolium</i>	
<i>Lactuca serriola</i>	<i>Torilis arvensis</i>	
<i>Stachys officinalis</i>		
<i>Verbascum spp.</i>	Rural/peri-urban farms	
<i>Vicia spp.</i>	<i>Cosmos bipinnatus</i>	
	<i>Lavendula spp.</i>	
Peri-urban farms only	<i>Trifolium spp.</i>	
<i>Alcea spp.</i>		
<i>Calendula officinalis</i>	Rural/urban farms	
<i>Celosia spp.</i>	<i>Origanum vulgare</i>	
<i>Gnaphalieveae</i>		
<i>Gomphrena globosa</i>	Peri-urban/urban farms	
<i>Hibiscus spp.</i>	<i>Achillea millefolium</i>	
<i>Lagerstroemia spp.</i>	<i>Anethum graveolens</i>	
<i>Limonium sinuatum</i>	<i>Centaurea cyanus</i>	
<i>Phlox spp.</i>	<i>Convolvulus arvensis</i>	
<i>Tagetes spp.</i>	<i>Gaillardia pulchella</i>	
<i>Tanacetum parthenium</i>	<i>Helianthus spp.</i>	
<i>Verbena bonariensis</i>	<i>Leucanthemum vulgare</i>	
	<i>Plantago spp.</i>	
	<i>Pycnanthemum muticum</i>	

Effects of habitat type

No significant differences were observed in pollinator abundance or species richness between habitat types (Fig. 2.3a, 2.3b). However, urban farms and prairies generally had more pollinators than the other farm types, and peri-urban farms generally had greater species richness than the other site types.

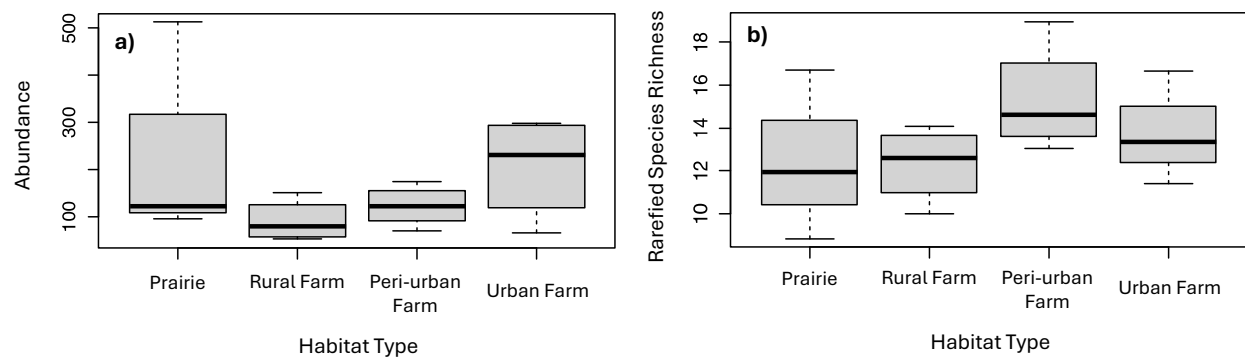


Figure 2.3. Boxplots of (a) pollinator abundance and (b) rarefied pollinator species richness by habitat type. No significant differences were observed between any groups.

PCoA and pairwise PERMANOVA of the unweighted pollinator communities (presence/absence) revealed that rural farms had a significantly different composition of species from prairies ($p=0.04$) and urban farms ($p=0.03$; Fig. 2.4). Rural farms had a greater association with *Epimelissodes obliquus* and *Melissodes bimaculatus*. Urban farms were associated with the *Hylaeus mesillae* species complex, *Lasioglossum imitatum*, *Stelis louisae*, and *Halictus rubicundus*. Prairies were associated with *Eristalis transversa* and *Eristalis stipator*.

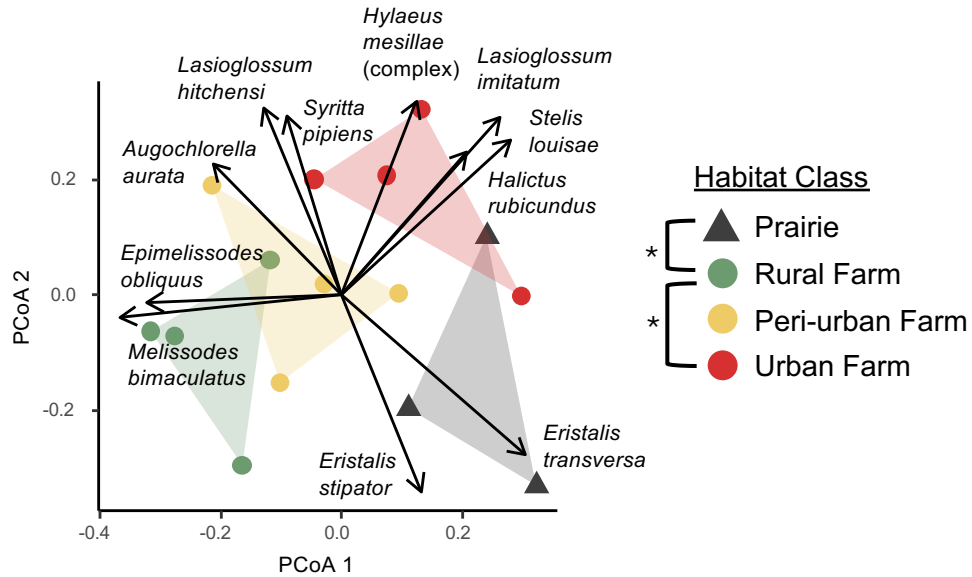


Figure 2.4. PCoA of unweighted (presence/absence) pollinator communities. Pairwise PERMANOVA results showed significant differences between rural farms and prairies ($p=0.04$), and rural farms and urban farms ($p=0.03$).

An ANOVA test of floral species richness showed no significant differences between habitat types (within farms or between farms and prairies; $p=0.138$; Fig. 2.5). However, urban and peri-urban farms tended to have more flowering plant species than rural farms or prairies.

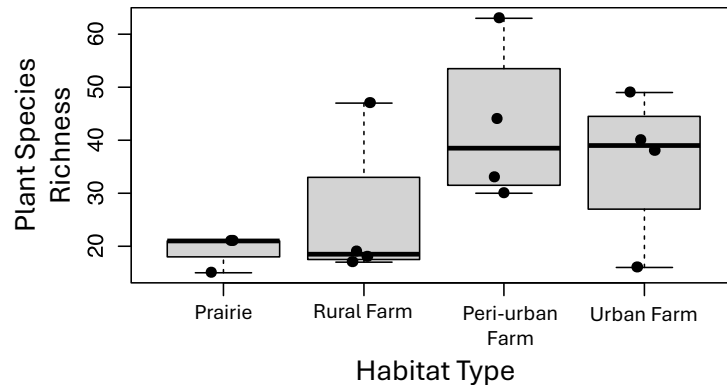


Figure 2.5. Boxplot of flowering plant species richness by habitat type. No groups were significantly different from each other according to an ANOVA test ($p=0.138$).

We also examined the plant community of our bipartite network using PCoA and pairwise PERMANOVA (Fig. 2.6). We could only test the unweighted community

(presence/absence), as we did not have the total abundance of plants for each site. This analysis showed that prairies had a significantly different plant composition than any type of farm (urban, $p=0.05$; peri-urban, $p=0.04$; or rural, $p=0.03$). Rural and peri-urban farms were associated with *Trifolium spp.*, urban farms were associated with *Echinacea spp.*, and prairies were associated with a variety of wildflower species, including *Pycnanthemum tenuifolium*, *Dalea purpurea*, *Eryngium yuccifolium*, *Liatris spp.*, *Silphium integrifolium*, and *Ratibida spp.*

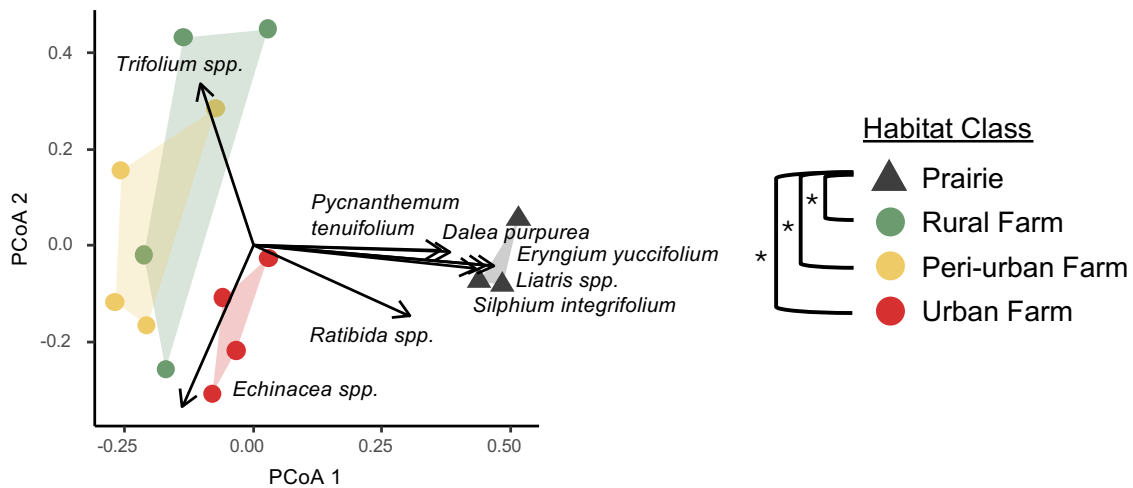


Figure 2.6. PCoA of the unweighted (presence/absence) plant community from observations. Pairwise PERMANOVA showed significant differences between prairies and urban ($p=0.05$), peri-urban ($p=0.04$), and rural farms ($p=0.03$).

We found no significant differences in modularity or network-level specialization z -scores across the different habitat types (Fig. 2.7). However, farms varied in specialization and modularity compared to the null model where all rural farms were more specialized and modular than expected by chance (denoted by red points in Fig. 2.7). These results suggest that the behavior of pollinators on rural farms was different from random. In contrast, the other habitat types (urban, peri-urban, and prairies) had some sites that were not significantly different from the null expectations for modularity and network specialization (denoted by black points in Fig.

2.7).

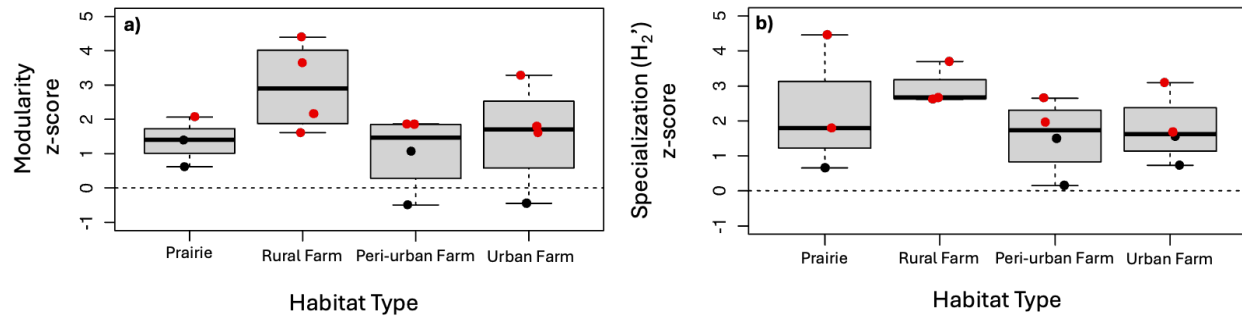


Figure 2.7. Boxplots of modularity and network specialization (H₂') z-scores by habitat type. Red points indicate a significant difference ($p < 0.05$) from the null model (significantly different from 0).

We compared the z-scores of species level specialization (d') between farm types and prairies for 18 pollinator species: *Apis mellifera*, *Bombus* spp., *Lasioglossum imitatum*, *Ceratina mikmaqi*, *Augochlorella aurata*, *Halictus ligatus*, *Lasioglossum hitchensi*, *Toxomerus marginatus*, *Melissodes bimaculatus*, *Eristalis arbustorum*, *Eristalis stipator*, *Ceratina strenua*, *Paragus haemorrhous*, *Lasioglossum trigeninum*, *Epimelissodes obliquus*, *Augochlorella persimilis*, *Xylocopa virginica*, and *Halictus confusus* (Fig. 2.8). Each of these species had at least 18 individuals and was found at a minimum of three habitat types. Significant differences were found in ANOVA tests between habitat types for only one species: *Lasioglossum trigeninum* ($p = 0.05$). Tukey's HSD test found that this species was significantly more specialized on rural farms than urban farms (Fig. 2.8n). It should be noted that this species was only found at one rural site.

Most species were not significantly more or less specialized compared to the null expectation (z-score of zero). *Eristalis arbustorum*, a species of syrphid fly, however, was significantly more specialized than expected at all urban and peri-urban sites it was found at (Fig.

2.8j).

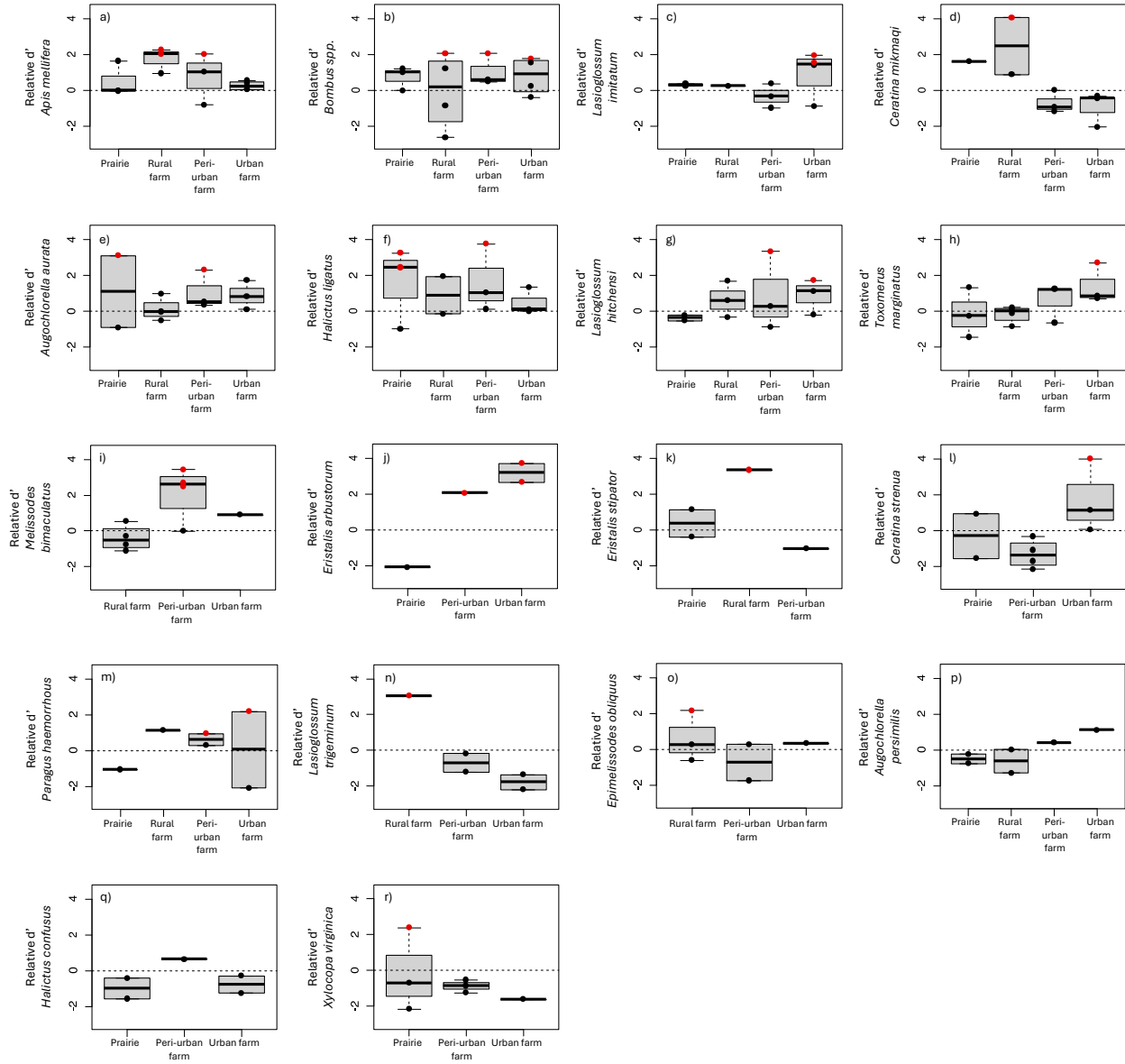


Figure 2.8. Boxplots of species specialization (d') z-scores by habitat type for select species. Red points denote a significant difference from the null model (significantly different from zero).

Effects of site size and floral richness

To determine whether pollinators were responding to site-specific features, we examined relationships between farm size and floral richness on pollinator structure and functional

behavior. We found no significant relationships between farm size and pollinator abundance, richness, modularity, or network specialization (Table 2.5). In contrast, floral richness negatively affected modularity ($\beta = -0.91$ $p=0.02$; Fig. 2.9a), and network specialization ($\beta = -0.79$ $p=0.03$; Fig. 2.9b), but it did not correlate with pollinator abundance or richness (Table 2.5).

Table 2.5. Results of linear models for pollinator abundance, pollinator richness, modularity, and network specialization compared to total floral richness and site size. Standardized beta values and their corresponding p-values are given. Adjusted R^2 values are also provided for each response variable.

	Pollinator abundance	Pollinator richness	Modularity	Network specialization (H_2')
Floral richness	$\beta = -0.22$ $p=0.46$	$\beta=0.56$ $p=0.07$	$\beta = -0.91$ $p=0.02^*$	$\beta = -0.79$ $p=0.03^*$
Site size	$\beta = -0.50$ $p=0.11$	$\beta=0.49$ $p=0.10$	$\beta = -0.49$ $p=0.18$	$\beta = -0.53$ $p=0.12$
Adjusted R^2	0.06	0.17	0.25	0.26

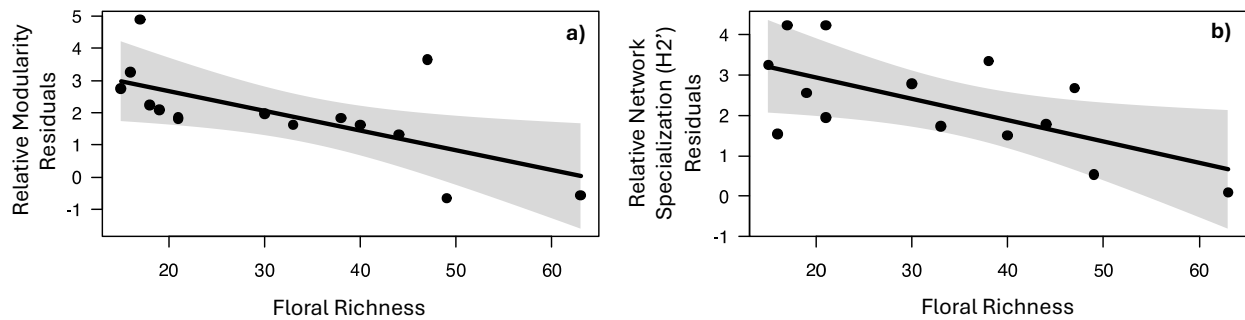


Figure 2.9. Residual regressions of (a) relative modularity (partial $R^2 = 0.36$, $p=0.02$) and (b) relative network specialization (partial $R^2 = 0.36$, $p=0.03$) compared to floral richness.

We also compared species specialization (d') z-scores across floral richness and site size gradients using the same species as previously analyzed. None of these species significantly

altered their foraging behavior across our size gradient. Only two species, *Ceratina mikmaqi* and *Augochlorella persimilis*, significantly altered their foraging behaviors along the gradient of floral richness ($p=0.03$, $p=0.04$, respectively; Fig. 2.10). *C. mikmaqi* specialization was negatively correlated with floral richness (more generalized at sites with high floral richness; $\beta=-0.09$, $p=0.03$; Fig. 2.10a). Interestingly, *A. persimilis* showed the opposite trend, and its specialization was positively correlated with floral richness (more generalized at sites with low floral richness; $\beta=0.05$, $p=0.04$; Fig. 2.10b).

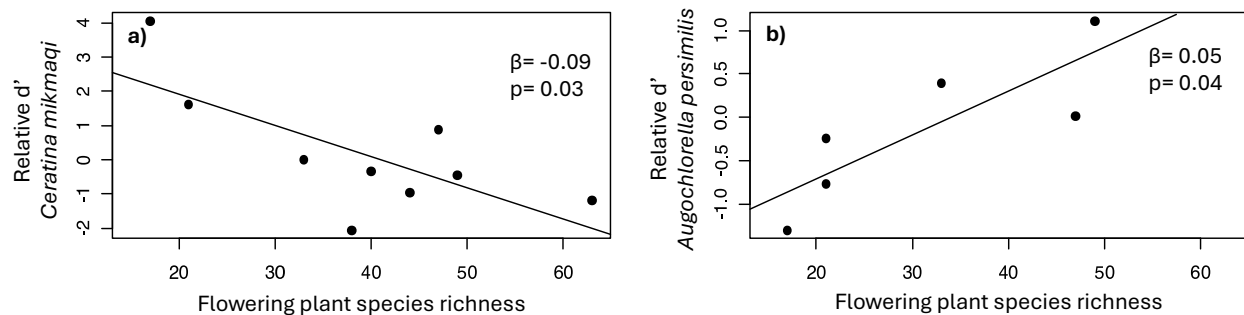


Figure 2.10. Regressions of relative species specialization (d' z-scores) of (a) *Ceratina mikmaqi*, and (b) *Augochlorella persimilis* across sites of different floral richness. Zero indicates the null expectation.

Discussion

The goals of this study were to investigate the effects of urbanization, habitat type, floral richness, and site size on the pollinator community. We did this in a variety of ways, examining pollinator abundance, richness, community composition, modularity, and specialization. We found that pollinator abundance and richness were unaffected by the variables we tested. Pollinator species assemblages differed based on habitat type and urbanization level. Networks tended to be specialized, and the degree of specialization was negatively related to local floral richness. However, the specialization of individual species was predominantly unaffected by any

local or landscape variable, except for two species, which altered their foraging behavior depending on floral richness.

Pollinator abundance and species richness

We hypothesized that pollinator abundance would positively correlate with urbanization on farms, and that prairies would have more pollinators than farms. However, we found no differences in abundance along an urbanization gradient or between habitat types. We also examined how local variables like floral richness and site size affected the abundance of pollinators and predicted that abundance would positively correlate with these variables. These results also showed no significant correlations. Pollinator richness also did not correlate with any of the predictor variables we tested (habitat type, floral richness, or site size). Other variables like floral abundance, planting density, and phenology may be more influential in shaping pollinator abundance and species richness (Neumüller et al., 2021; Plascencia & Philpott, 2017).

Pollinator community composition

Urban and rural farms were expected to have different pollinator compositions, which we found when examining unweighted pollinator species composition. We had also expected rural farms to have the most similar compositions to prairies, which was not the case. Prairies incidentally had significantly different compositions from rural farms and were more similar to urban and peri-urban farms. Approximately half of the pollinator species found at each type of farm were shared with prairies. We believe that because rural farms had the lowest number of species, they were considered significantly different, despite sharing the same proportion of pollinator species with prairies as other types of farms.

Urban farms and rural farms also had significantly different (unweighted) pollinator species compositions. Literature supports this finding, as certain species are known to be more adapted to urban environments (Bates et al., 2011)

Network modularity

We expected rural farms to have more modular networks than urban farms, and prairies to be more modular than any type of farm. However, we saw no differences in modularity on farms across the urbanization gradient or between farms and prairies. This suggests that these pollinator networks may be equally resilient to disturbance(s).

After finding no significant differences between groups, we looked for patterns of modularity being significantly different from the null model expectations (zero). Rural farms were consistently more modular than expected, but all other site types (prairies, peri-urban farms, and urban farms) had some modularity scores that were not significantly different from the null expectation. This indicates that foraging preference is occurring at rural farms, as clear modules (sub-groups of plants and pollinators interacting together) were being formed more frequently than at random. This may be indicative of resource partitioning occurring in these systems (Cordeiro et al., 2020). Resource partitioning is more common in networks that are more specialized, which we also observed in our study. Thus, this pattern in modularity we see at rural farms is likely due to the pollinators being more specialized in this type of habitat.

We also hypothesized that modularity would positively correlate with floral richness and site size. Although we found no relationship with site size, our results showed that modularity negatively correlated with floral richness. This suggests that enhancing floral richness may lead to greater niche overlap for pollinators. In a system like this, specialist pollinators visit a subset

of the plant species that generalists visit (Bascompte et al., 2003). With greater niche overlap, we might also expect to see more generalist foraging occurring (less specialization), which did occur (Table 2.5). Floral richness explained 25% of the variation observed in modularity, which is relatively low. It is likely that other variables are also important predictors of this metric. For instance, flowering phenology and habitat loss have been found to influence network modularity in other studies (Spiesman & Inouye, 2013; X. Wang et al., 2025).

Network and species-level foraging behavior (H₂' and d')

We hypothesized that pollinator networks at rural farms would have the most specialized foraging patterns of all farms, and prairie pollinator networks would be the most specialized overall. However, we found no significant relationship between network specialization (H₂') and urbanization on farms, or between farms and prairies. Network specialization was also not significantly correlated with site size, which was contrary to our hypotheses (we had predicted a positive correlation between specialization and site size). However, we did find a negative correlation between network specialization and floral richness, which was contrary to our hypothesis. This relationship may have been tied to the abundance of floral resources. At sites with high floral richness, there were fewer amounts of any one kind of flower compared to sites with low floral richness. This would mean that specialists would have reduced access to resources, favoring generalization. Floral richness explained 26% of the variation observed in network specialization, which is relatively low. Other unmeasured variables like floral abundance and phenology also likely contributed to the differences we observed (Souza et al., 2018).

Species-level specialization (d') values suggested that individual pollinator species also generally had similar foraging behaviors regardless of the habitat type in which they were found. Foraging behavior within a species generally did not differ based on the size of the site or total floral richness. This would suggest that most species tend to keep similar dietary breadths regardless of these contextual differences in the environment. However, we did find that two bee species, *C. mikmaqi* and *A. persimilis*, altered their diet breadths depending on the local floral richness (Fig. 2.10). These species responded in opposite ways; as floral richness increased, *C. mikmaqi* became more generalized, and *A. persimilis* became more specialized. These results suggest that different pollinator species use different foraging strategies and may respond in unique ways to local variables.

Conclusions and conservation implications

We found that although species composition differed between habitat types, pollinator abundance, richness, modularity, and specialization were relatively unaffected by these variables. Therefore, some farms and prairies may function similarly, but they do not necessarily support the same pollinator species. This is an important consideration when determining conservation measures. If the goal is to restore ecosystem function (i.e., network structure) to a landscape, farms and prairies may be interchangeable as habitat types. However, if the goal is to support specific target species, these two habitat types may not be interchangeable.

We also found that floral richness impacted pollinator foraging behavior at the network level (and species level for a few bees) and was important for pollinator network structure. This local variable is far easier to control than landscape factors like surrounding developed cover,

which is encouraging for conservation efforts. Increasing floral richness should be considered to enhance the quality of habitat for pollinators regardless of landscape context.

Future research should explore additional variables such as floral abundance and floral phenology, as these have been shown to influence pollinator networks (Russo et al., 2013; Yahaya et al., 2024). Additionally, it would be beneficial to examine the effects of frequent planting rotations and tillage on bee species compositions at farms (Kratschmer et al., 2018; Ullman et al., 2020).

This study only sampled a few prairie sites; however, to our understanding, there are few, if any, projects that compare pollinator networks at restored prairies to those of vegetable farms across a gradient of urbanization. Therefore, we find value in the novelty of this study and suggest that future studies continue to compare pollinator networks on farms to those of prairies to better understand differences between these habitat types. This study was also only conducted during one season (summer 2025). Climatic differences between years may influence patterns in arthropod communities. Our study took place in the Kansas City Metropolitan area; this is an area within the central US where little is known about pollinator ecology. All cities have unique layouts, differing in infrastructure densities, amount of green space, and connectivity between green spaces. Therefore, the patterns of arthropods in one city may not be entirely applicable to another. More research should be conducted to compare pollinator community structure and function between different cities. As cities continue to expand and urban agriculture becomes more popular, it will become increasingly important that we understand how best to support pollinators. Our study contributes to this knowledge base and suggests directions for future investigations to explore.

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Conclusion

In our first study, we found that urbanization did not consistently affect arthropod abundance or community composition. Instead, local variables like crop richness (and potentially floral abundance and management practices like tilling or mulching) seem to be more important drivers of these metrics. Overall, these findings are promising for insect conservation, as local variables are far easier to manage than landscape conditions. Landscape modification via urbanization is nearly inevitable, but it may not be as detrimental to arthropod populations as previously thought. We also found that farms supported similar or greater quantities of beneficial arthropods as prairies, suggesting that farms and prairies may be equivalent habitat types for these arthropods.

In our second study, we examined the pollinator community specifically to understand how habitat type (i.e., urban farm, peri-urban farm, rural farm, prairie), floral richness, and site size may have affected foraging behavior and community stability. Our results showed that these variables also generally were not important drivers of foraging behavior and community stability in pollinators. We believe that other local variables, like floral abundance, were also important predictors of these metrics. In this study, we also revisited the comparison between farms and prairies to see if any patterns emerged on a finer scale. Farms and prairies did not necessarily support the same pollinator species, but did exhibit similar patterns in pollinator foraging and community stability. This suggests that farms may maintain similar ecological function to prairies, despite harboring different species compositions.

Future studies should examine other local and landscape variables to gather a better understanding of which environmental factors arthropods are most sensitive to. We suspect that variables like habitat connectivity, soil management practices, and floral abundance contributed

to some of the variation we found in arthropod community abundance, composition, and behavior. This work should be conducted to determine which variables are the most important to consider when introducing new green spaces in urban areas, or how best to manage current green spaces for the benefit of arthropods. More research should also be done to compare arthropod communities across a gradient of urbanization in prairies and compare this to farms across a similar gradient to understand whether the effects of developed cover may influence arthropods differently in different habitat types. Different habitat types may be more conducive to arthropod community stability in the face of urbanization and its coinciding anthropogenic stressors. Additionally, farms should be compared to other types of green spaces (i.e., city parks, arboretums, cemeteries, vacant lots, bioswales) to further understand the value of UA as habitat for beneficial arthropods and determine which types of green space may be the most beneficial to incorporate in urban spaces from a conservation standpoint.

Overall, this project affirms prior claims that vegetable farms serve as beneficial habitats for arthropods. Thus, incorporating more UA in cities may be one avenue to mitigate local arthropod declines. Our findings also highlight the importance of local variables, such as floral/crop richness, in shaping arthropod communities. Local manipulation of habitat quality (i.e., planting more floral resources/more diverse flowering plants) may be more important in some cases than the landscape context in creating “arthropod-friendly” habitats.

This project also addressed a knowledge gap in the comparison of arthropod communities at farms and prairies. Our findings suggest that farms and prairies may be of similar conservation value as habitat for some beneficial arthropods, depending on conservation goals. For instance, if the goal is to restore a landscape’s ecological function (i.e., general community composition patterns, community stability, or plant-pollinator visitation patterns), adding organic farms may

be just as beneficial as adding more natural area. However, if the goal is to better support specific rarer groups of beneficial insects (i.e., rare pollinator species), farms may not have the same habitat value as prairies.